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PHYSICAL, CHEMICAL AND BIOLOGICAL EFFECTS OF DREDGING IN THE TH--ETC(U)
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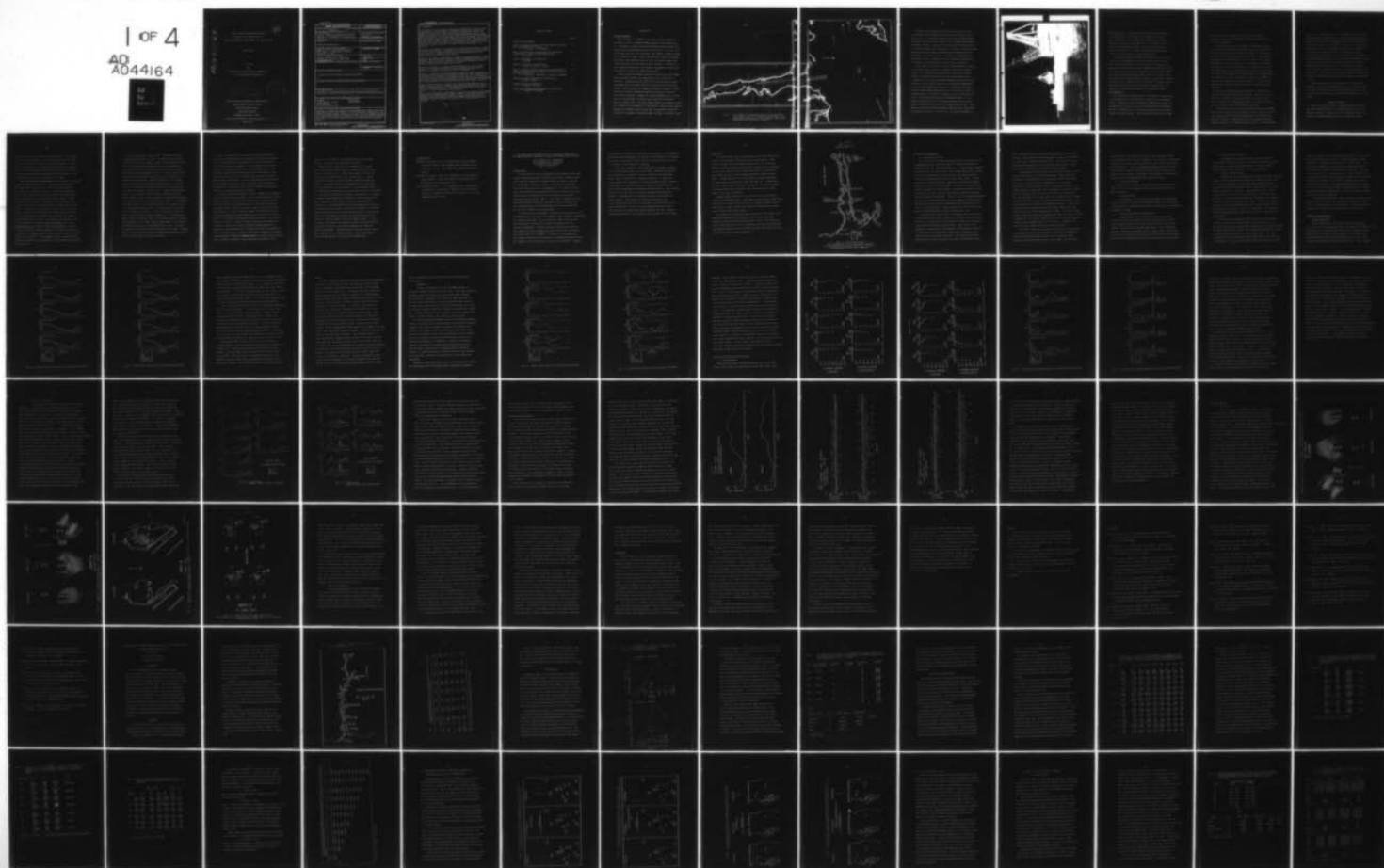
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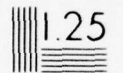




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Physical, Chemical and Biological Effects
of Dredging in the Thames River (CT)
and Spoil Disposal at the New London (CT) Dumping Ground

Final Report

To

U. S. Navy

and

Interagency Scientific Advisory Subcommittee
on Ocean Dredging and Spoiling

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April 1977

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Field surveys of the Thames River hydrography, phytoplankton, and trace metal concentrations in water, sediment and shellfish suggested that effects of dredging on primary production were spatially and temporally limited. The highest concentrations of nickel, lead, cadmium and mercury in water samples were observed before or during dredging, while copper was highest after dredging but were generally higher upriver. Sediment levels of these five metals, plus zinc and organic carbon, increased in an upriver direction. Dredging-related changes in trace metal body burdens in shellfish but were difficult to separate from normal seasonal variations. No gross pathology was detected in the shellfish.

Studies on the chemical oceanography of disposal area showed that water quality parameters (Eh, pH, turbidity, dissolved oxygen, suspended and volatile solids) showed no patterns relative to distance or direction from dumpsite, though seasonal changes in these parameters were evident.

Studies on sediments and benthic macrofauna showed that surface sediments at the disposal point and several surrounding stations ($\leq \frac{1}{2}$ n mi from this point) clearly had large increases in percentages of silts and clays, indicative of the presence of spoils, during and after spoiling. No spoil impacts, distinguishable from apparently natural fluctuations, were detected outside of the pile.

Scuba survey and underwater photography of disposal site indicated that troughs mounds of newly dumped material provided relief to the normally flat bottom topography and were initially attractive to demersal fish and crustacea. No obvious turbidity was noted at the spoil-water interface during 1.5 knot currents.

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INTRODUCTION

Dredging Background

The U. S. Navy, to accommodate a new class of nuclear submarines at its Groton, CT, base, is dredging extensive portions of the Thames River, CT. The dredging is largely new work which deepens existing channels and berthing areas from predredging depths of 32-33 ft (9.8 - 10.1 m) to 36 ft (11.0 m) plus one foot (0.3 m) overdredge. The dredging is being conducted in two increments or phases. Phase I extended from 19 August 1974 through 18 July 1975, and included the portion of the river channel between Long Island Sound and the Gold Star Memorial Bridge (Figure 1). According to U. S. Army Corps of Engineers (COE) estimates, approximately 1,510,112 yd³ (1,156,746 m³) of spoils were removed during Phase I.

Future planned Navy dredging includes the section of the river channel from the Gold Star Bridge to the submarine base (ca. 1,292,000 yd³ or 989,672 m³). Another 195,000 yd³ (149,370 m³) of spoils would be created in extending the river channel about 533 ft (163 m) north of its present terminus. Berthing areas at the submarine base must be dredged to -39 feet (11.9 m) with one foot overdredge, generating an additional 514,000 yd³ (393,724 m³) of materials. Dredging of a floating drydock area to -59 feet (18.0 m) is also planned; this would involve 154,000 yd³ (117,748 m³) (Dept. of the Navy, 1976). The total volume of spoils to be removed in Phase II is estimated to be 1,845,000 yd³ (1,410,687 m³). At this writing, Phase II dredging was scheduled to begin in June 1977. An additional 200,000 yd³ (152,920 m³) of dredging is projected through 1985 (Dept. of the Navy, 1976).

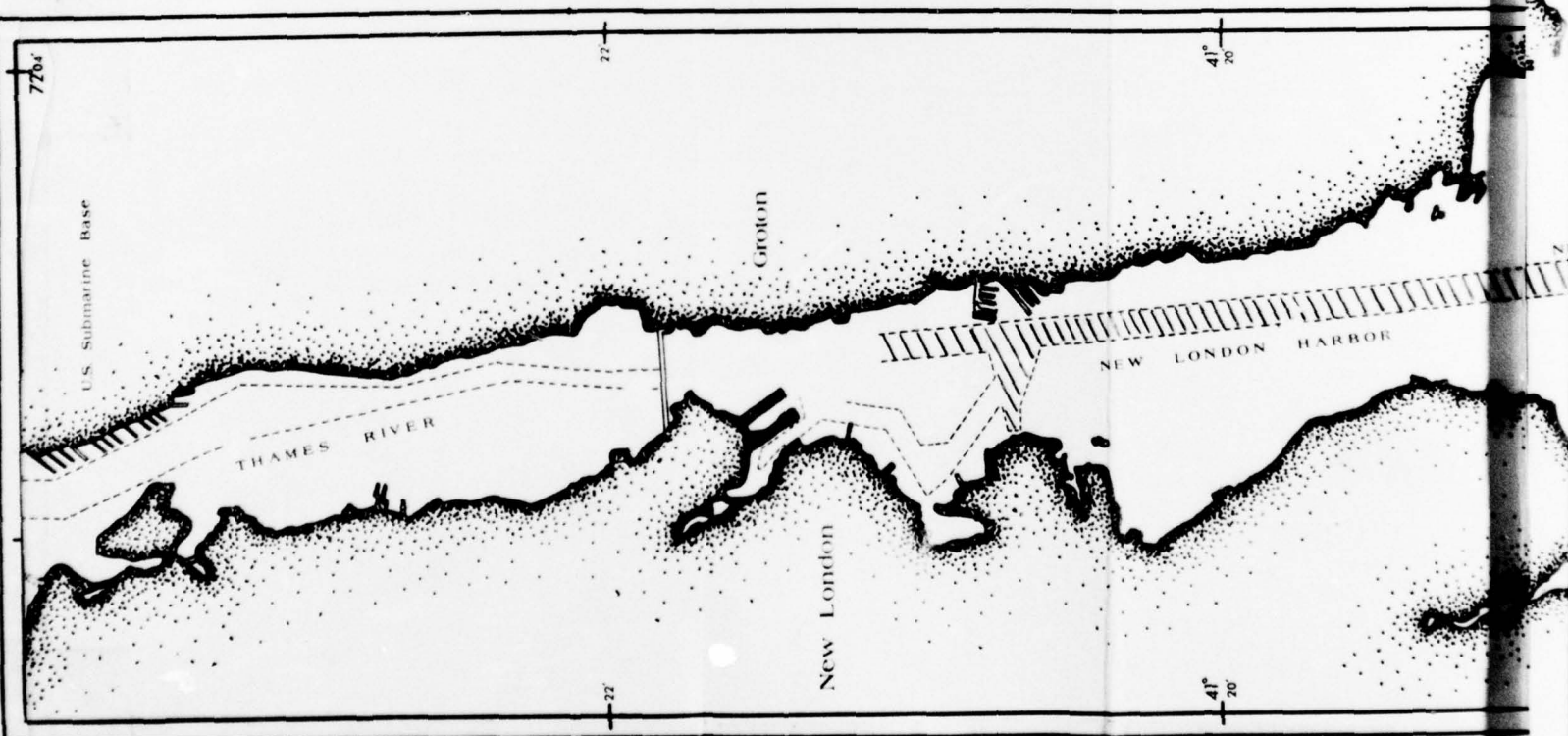
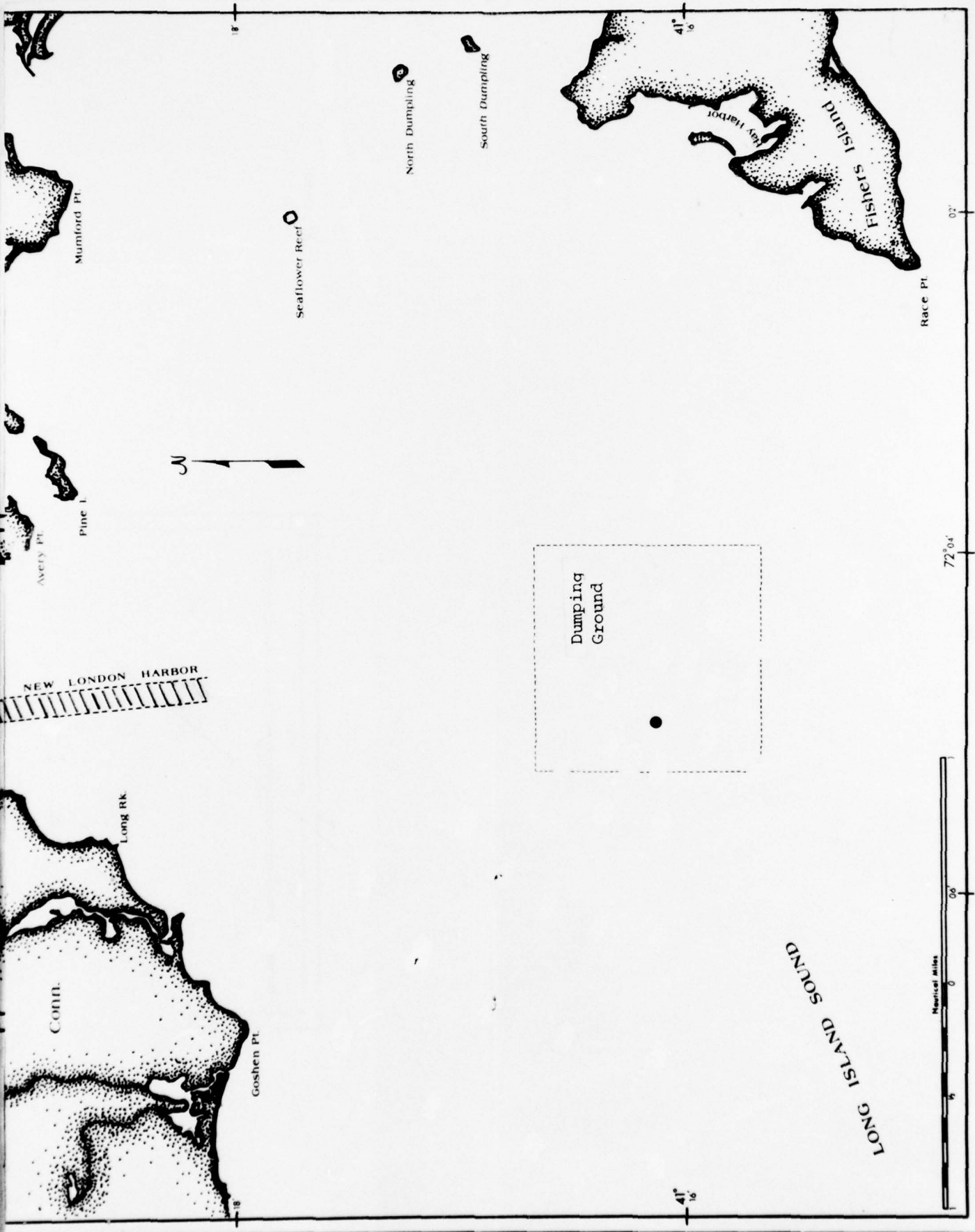


Figure 1. Lower Thames River, showing approximate area dredged during first increment (\\\\\\\\), and Eastern Long Island Sound, with designated New London Dumping Ground indicated. ● = Original disposal point for first spoil increment.



Sediments of the dredging areas have been described as uniformly clayey organic silts with moderate plasticity and cohesiveness (Dept. of the Navy, 1973). The sediments removed in the first increment exceeded interim EPA guidelines only for volatile solids, chemical oxygen demand and Kjeldahl nitrogen. Values for these indices decreased both in a downriver direction and with depth in the sediments. That most spoils deposited to date have been fairly cohesive and uncontaminated must be considered in evaluating this report's findings on contaminant levels and impacts on biota. Sediments still to be removed are considered less consolidated (at least surficially) as well as more contaminated.

Large capacity (10-14 yd³ or 7.7-10.7 m³) bucket dredges, and bottom-release scows (Figure 2) have been utilized in an effort to minimize turbidity in the dredging and disposal areas. Spoils from the first increment of dredging were dumped at the New London Dumping Ground, a 1 nautical mile (n mi) square designated disposal area bounded by 41°15.7' and 41°16.7' N, and 72°04.0' and 72°05.3' W, and centered approximately 2.7 n mi S of the Thames' mouth (Figure 1). Predisposal water depths within the dumping ground ranged from 47-81 ft (14.3-24.7 m) except for a mound reaching to 36 ft (11.0 m), probably a relict of past spoil disposal, in the north-central portion of the dumping ground. As a rule, depths increased from N to S, reflecting the general trend of deeper waters as one moves from the mouth of the Thames toward the Race. Point dumping of the spoils has been attempted, utilizing a buoy installed for this purpose at 41°16.13' N, 72°05.02' W, in the southwest quadrant of



Fig. 2. Dredge and barge employed in
Thames River dredging. 1975.

the dumping square. Spoils were released at this buoy from 19 August to 20 December 1974. At that time, due to formation of a peak of spoils, water depths at the disposal buoy had decreased from their original 70 ft (21.4 m) to less than 50 ft (15.3 m) depths. The disposal point was therefore shifted ca. 600 ft (183 m) to the SE, to 41°16.1 N, 72°04.99' W. On 22 July 1975, the buoy itself was moved ca. 150 ft (45.8 m) to the S (41°16.90 N, 72°05.00 W).

The New London Dumping Ground has historically served as a disposal area, receiving approximately 6,000,000 yd³ (4,656,000 m³) of spoils over the past 20 years (Dept. of the Navy, 1976). The largest recent addition was the 92,500 yd³ (70,855 m³) dredged by the Navy from a Groton pier construction site in 1972. Environmental effects of that operation were monitored (Naval Oceanographic Office, 1973) in an attempt to predict impacts of the larger project now underway. Several smaller dredging-disposal projects, including 14,500 yd³ (11,107 m³) by the Mystic Marine Historical Association and 7,400 yd³ (5,668 m³) by Amerada Hess, were carried out concurrently with the Navy's first dredging increment.

Study Background

Under the terms of the dumping permit, the dredging and disposal were to be accompanied by an extensive survey to monitor their environmental effects. The National Oceanic and Atmospheric Administration (NOAA) was designated the lead agency in these investigations, which were funded by the Navy. Coordination of all studies was undertaken by the Northeast Fisheries Center (MACFC), National Marine Fisheries Service, NOAA.

This group was also responsible for two specific investigations: microbiology (Section E of this report) and benthic macrofauna (Section F) of river and dumpsite.

The disposal permit also stipulated that a representative segment of the scientific and academic institutions of the Long Island Sound region should participate in the monitoring studies. On this basis, the New York Ocean Science Laboratory (NYOSL), Montauk, N. Y., was awarded subcontracts to investigate disposal impacts on physical and chemical oceanography, and finfish populations, of the dumpsite and surrounding areas (Sections C, D and G, respectively). The University of Connecticut (UCONN), Groton, Conn., was designated to determine effects of dredging on river flow characteristics and concentrations of suspended materials (Section A), as well as heavy metals in river water, sediments and shellfish, and shellfish gross pathology (Section B). UCONN has also conducted SCUBA surveys of the spoil pile and its lobster populations (Section H).

All the above investigations began with "baseline" surveys in June-July 1974, shortly before the onset of dredging. Sampling for most projects was on a quarterly or more frequent basis, extending at least through February 1976. Seven quarterly reports on activities and findings of the various projects were submitted to the Navy.

In August 1974, a set of criteria specifying the limits of acceptable impacts at the disposal area was released by COE after consultation with the Interagency Scientific Advisory Subcommittee on Ocean Dredging and Spoiling, or ISASODS. (ISASODS consists of one northeastern area representative from each of the following agencies: COE, U.S. Fish and Wildlife

Service, U. S. Geological Survey, U. S. Environmental Protection Agency and National Marine Fisheries Service. The representatives advise their respective regional directors on matters related to dredging and spoiling in northeastern waters. As of January 1977, the States of New York and Connecticut have also been made members of ISASODS.) The majority of these monitoring criteria considered a circle of one mile radius, centered at the disposal buoy, as the area within which some impacts were unavoidable and acceptable. If the criteria were exceeded, or other significant impacts detected, the monitoring study was to notify COE through ISASODS, and operations were to be halted until alternative, acceptable means of dredging and disposal, or an alternate disposal site, were agreed on. Where feasible, the monitoring surveys have adapted their objectives, sampling strategies and schedules to address the monitoring criteria.

Other studies related to the disposal project include several bathymetric surveys of the disposal area by COE and by the Naval Underwater Systems Center (NUSC), and current measurements by NUSC. These studies will be discussed only as they pertain to portions of the present report.

SUMMARY OF FINDINGS

A. Impact of Dredging Operations on Suspended Material Transport in the Lower Thames River Estuary: The suspended material field in the Thames River Estuary is characteristic of a sediment-poor system and displays low average concentration levels (<5 mg/l) and erratic

variability with no obvious spatial or temporal trends. In this system the Phase I dredging operation: 1) Produced perturbations in suspended material concentrations and composition that were confined to an area within 300 to 500 yards of the operating dredge and barge. 2) Produced an increase in total suspended load within the estuary that was small in comparison to that produced by typical aperiodic storm events. 3) Caused no major alterations in mass transport within the estuary.

B. Thames River Hydrography, Phytoplankton, and Trace Metal Concentrations in Water, Sediment and Shellfish: 1) Field surveys and elutriate experiments suggested that effects of dredging on primary production were spatially and temporally limited. 2) Highest concentrations of nickel, lead, cadmium and mercury in water samples were observed before or during dredging, while copper was highest after dredging. Nickel, lead and copper were generally higher upriver; mercury was more concentrated in the lower river. Mercury decreased in dredge-induced turbidity plumes. 3) Sediment levels of these five metals, plus zinc and organic carbon, increased in an upriver direction. Postdredging concentrations of zinc and copper were significantly reduced in lower river sediments in comparison with earlier values. 4) Dredging-related changes in trace metal body burdens in the oyster, Crassostrea virginica, hard clam, Merchenaria mercenaria, and morrhua clam, Pitar morrhuana were difficult to separate from normal seasonal variations. The significant decreases in five metals in C. virginica probably reflected a general reduction of these metals in the environment. In M. mercenaria, however, as many metals had increased concentrations as had decreases. No gross pathology was detected in the shellfish.

C. Physical Oceanography of Disposal Area: 1) Turbidity was higher in bottom waters than near the surface. This was not restricted to the vicinity of the spoil pile; however, slightly higher turbidity downstream of the pile indicated that some scouring of spoils was taking place.

2) The average speed of the turbidity or density cloud resulting from barge discharges was approximately 16 m/min. 3) Droque experiments and current meter records showed net flows to the SE at the surface, E at mid-depth and NE near bottom. Maximum transport was in the east/west direction with highest values occurring during the ebb. 4) A reversal from SE to NE flow appeared approximately midway in the ebb tidal cycle.

D. Chemical Oceanography of Disposal Area: 1) Water quality parameters studied (Eh, pH, turbidity, dissolved oxygen, suspended and volatile solids) showed no patterns relative to distance or direction from dumpsite, though seasonal changes in these parameters were evident. 2) Concentrations of Kjeldahl nitrogen, total phosphorus and seven metals (cadmium, copper, iron, nickel, lead, zinc and mercury) in seston were also not obviously related to proximity to spoiling. 3) Water quality rapidly (within two hours) returned to ambient following a dumping event. 4) Sediments from stations within or bordering on the 1 n mi² designated dumping ground were generally higher in iron, copper, chemical oxygen demand, total phosphorus and Kjeldahl nitrogen than were sediments outside the area. No evidence of spreading of the material outside of the dumpsite was found. 5) Benthic animals collected inside and outside the dumping ground had similar concentrations of heavy metals, phosphorus and nitrogen.

E. Effect of Dredging and Spoils Deposition on Fecal Coliform Counts in Sediments and Bottom Waters of the Thames River and New London Disposal Site: 1) Predredging fecal coliform densities were high (mean 14,000/100 ml) in surficial river sediments. 2) Spoil disposal did not increase densities in sediments and bottom waters of the disposal area. 3) No significant differences were found between fecal coliform counts in sediments within one-half n mi of the disposal point and counts at stations further removed from spoiling. 4) Counts in bottom waters of the spoiling area were higher on ebb tides than on flood tides. This indicates that the Thames River outflow plays a major role in determining fecal coliform densities in disposal area waters and sediments.

F. Sediments and Benthic Macrofauna: 1) Surface sediments at the disposal point and several surrounding stations ($\leq 1/2$ n mi from this point) clearly had large increases in percentages of silts and clays, indicative of the presence of spoils, during and after spoiling. Spoils were not detected at greater distances, perhaps due in part to sampling variability and sediment patchiness. 2) Species diversity, numbers of individuals and species of benthic macrofauna were greatly reduced within the spoil pile. No spoil impacts, distinguishable from apparently natural fluctuations, were detected outside of the pile. 3) Changes in species composition were somewhat more widespread. Populations of several predisposal dominant amphipods and bivalves decreased throughout the 2 n mi radius study area, and polychaetes at ≤ 1 n mi from spoiling. No changes were evident in numbers of the overall dominant species, the amphipod, Ampelisca vadorum. 4) Samples taken 2-14 months after termination of disposal showed progressive recolonization of the

spoil pile. In the absence of additional spoiling, a fairly complete recovery of the disposal area fauna would be expected.

G. Demersal Finfish of Disposal Area: 1) Numbers of fish caught dropped sharply after disposal began, and postdisposal catches did not return to the baseline level. It was thought, however, that the variation might be a seasonal rather than a spoiling effect. 2) Winter flounder was the species collected in greatest numbers throughout the study. 3) Gut content analysis showed windowpane flounder and tautog to be very selective in their food habits. The other abundant species were opportunistic feeders and had a varied diet, mostly benthic invertebrates.

H. SCUBA Survey and Underwater Photography of Disposal Site: 1) Troughs and mounds of newly dumped material provided relief to the normally flat bottom topography and were initially attractive to demersal fish and crustacea. 2) A forty-foot mound of previously-disposed spoils contained high concentrations of lobsters, crabs and winter flounder. 3) No obvious turbidity was noted at the spoil-water interface during 1.5 knot currents. Small (0.5-1.0 cm diameter) clay balls were observed rolling over the spoil pile; this may constitute a form of sediment transport. 4) Dives in May-June 1976 revealed that scouring of sediment and disintegration of clay mounds had occurred. The general topography of the mound had flattened considerably. There was evidence of sorting, leaving a cap of shell fragments and 2-3 m diameter patches of coarse gravel material. Spoils may have spread to 0.5 n mi SE of the disposal buoy.

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A. AN INVESTIGATION OF THE IMPACT OF DREDGING OPERATIONS ON
SUSPENDED MATERIAL TRANSPORT IN THE LOWER THAMES RIVER ESTUARY

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Introduction

In many areas of the northeastern United States harbor and channel dredging operations represent the major transport mode affecting the movement of sediment from the river or estuary to the adjacent continental shelf. Very often the bulk of these sediments have been contaminated by the variety of municipal and industrial discharges entering the coastal waters and contain high concentrations of both organic and inorganic pollutants (Boyd, et al., 1972). As a result, dredging and the attendant disposal operations generally tend to increase the areal distribution of these materials and their availability to the lower members of the food chain.

In recent years considerable effort has been directed to the determination of the impact of dredging induced sediment dispersion on coastal ecosystems. For the most part these studies have been primarily concerned with impacts associated with the disposal operations. Relatively little work has been done on the influence of the dredging operation itself (see the review by Morton, 1976, for example). To date most investigations have simply considered the esthetics of dredging placing particular emphasis on the reduction or control of turbidity. Limited

information is available describing the character of the sediments suspended during dredging, the spatial distributions of materials, concurrent contaminant release and the resultant short and long term impacts on the adjacent ecosystem.

In the early summer of 1974 initiation of channel dredging in the lower Thames River near New London, Connecticut provided an opportunity to examine in detail several of the most prominent impacts produced by a dredging operation. A series of field surveys were designed to examine the quantitative influence of dredging on suspended material transport, trace metal distributions and biological community characteristics. This section presents the field data and initial analyses describing the impact of dredging on the suspended material field within the lower estuary. Both direct and indirect influences are to be considered; the former by analyses of the material field in the immediate vicinity of the operating dredge and barge, and the latter through quantitative estimates of mass transport variations induced by the increased cross section of the estuary.

Study Area

The Thames River (Fig. 1) represents one of the three major streams contributing freshwater to Long Island Sound. This narrow, incised estuary extends sixteen miles in a northerly direction from its mouth in the eastern Sound to the confluence of the Yantic and Shetucket Rivers near Norwich, Connecticut and drains an area of approximately 400 mi². (Thomas, et al., 1968). During the past five years annual average streamflow of this system varied between 2500 and 2700 cfs. Comparisons of these levels with concurrent streamflows from the Housatonic and Connecticut Rivers indicate that the Thames River ranks third in relative importance.

Regular tidal variations can be observed over the entire length of the Thames River. Mean tidal range varies progressively from 2.6 Ft. at New London to 3.0 Ft. at Norwich (NOAA, 1976). Near surface tidal currents are variable and generally average less than 0.8 knot (NOAA, 1971).

The tidal and streamflow characteristics of the Thames River generally permit the intrusion of saline waters to extend upriver to Norwich (Soderberg and Bruno, 1971). The resultant density distributions favor a persistent vertical stratification that significantly affects both the velocity field and concurrent material transport within the river.

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THAMES RIVER
Station Locations

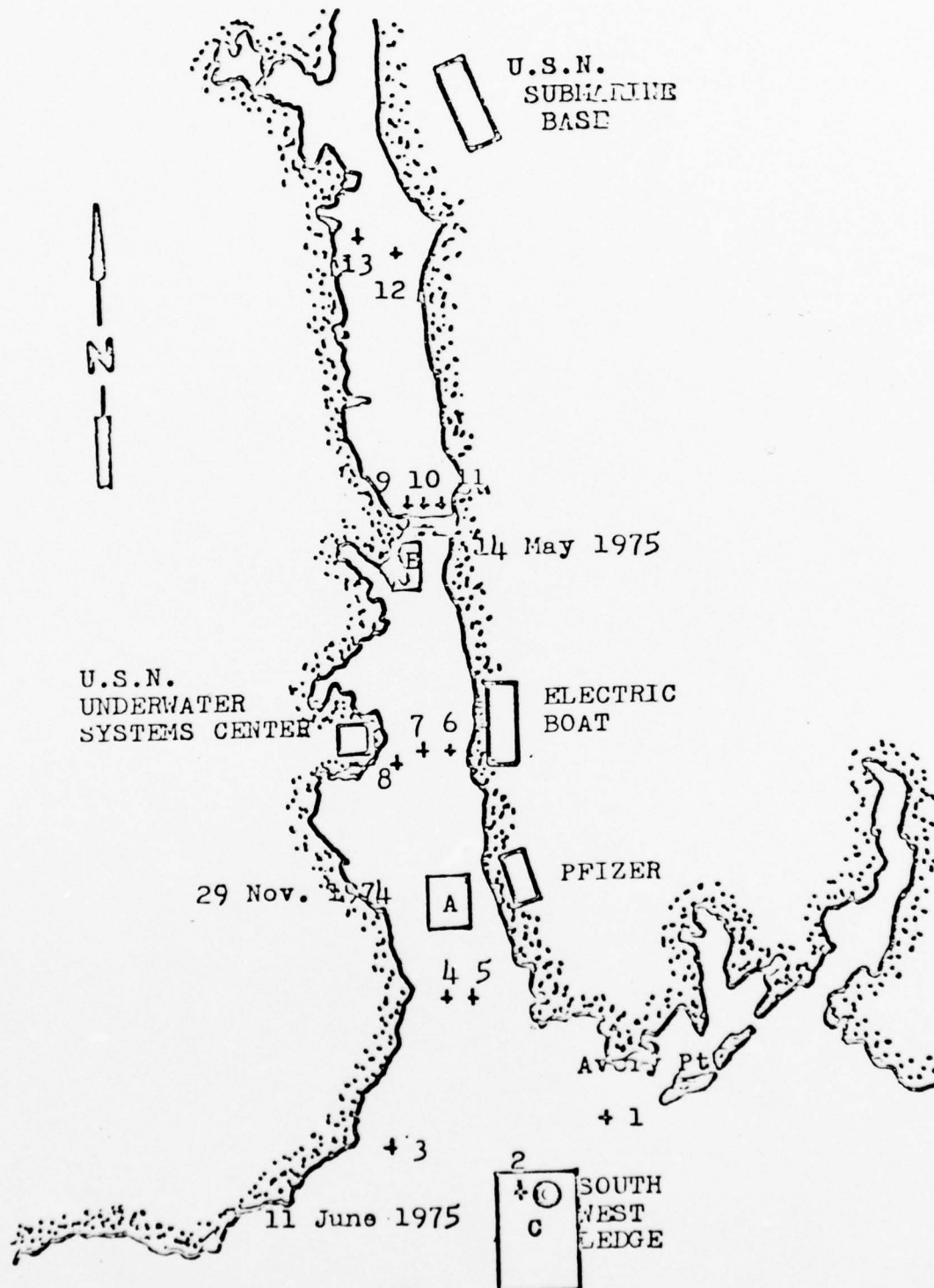


Fig. 1. Lower Thames River
Numbers designate stations sampled monthly.
Letters indicate locations where high resolution
dredge/barge surveys were conducted.

Methods and Procedures

a. Sampling Procedures

To determine the average background characteristics of the suspended material field a network of thirteen stations was established in the lower river (Fig. 1). Stations were sampled monthly during the period July, 1974 through May, 1976. Survey periods were confined to the ebb tidal cycle with sampling initiated approximately two hours after slack water. At each station, 4.0 liter drawn water samples were obtained at three locations on the vertical: surface, mid-depth and near bottom, using a standard Van Dorn sampler. Concurrent profiles of the vertical temperature gradient were obtained using an in situ probe (Hydrolab Corporation). During the first year, measurements of optical transmissivity were also obtained at several points in the water column using a Martek Model XMJ Transmissometer with a 0.25m pathlength. These measurements were discontinued after analyses indicated limited quantitative value.

To examine the direct impact of dredging operations on the suspended material field, a series of detailed high resolution surveys were conducted in the immediate vicinity of the operating dredge and attendant spoils scow. Each survey first employed transmissometer measurements to determine the spatial distributions of the plume of turbid water produced by the dredging and scow loading operation. Quantitative material distributions within the plume were then established by drawn

water sampling using the same methods employed in the monthly surveys. The number of stations sampled varied in each survey as a function of plume characteristics. The location of each station was referenced to the dredge and barge using a hand-held bearing compass and horizontal sextant angles.

To complement the measurements of suspended material concentrations and assist in the determinations of mass flux, the magnitude and variability of the velocity field in the lower river over a tidal cycle was examined on two occasions: December 12, 1974 and May 28, 1975. During the December, 1974 survey velocity measurements were obtained at a single location near the mouth of the river in the vicinity of navigational buoy R-6 (Station A). In May, 1975 this station was reoccupied concurrently with a second station located approximately 0.1nm (0.2km) south of the Gold Star Bridge (Station B). Each set of measurements was obtained from small boats moored along the center line of the main navigational channel. At each location, current speed and direction measurements were obtained at several locations on the vertical using a Bendix Q-15 ducted propeller current meter equipped with a deck read-out. Profiles were surveyed every fifteen minutes for thirteen hours.

During May, 1975, survey channel stations were supplemented by secondary survey grids designed to determine the degree of lateral variability characteristic of the velocity field. Stations were established to the north, south, east and west,

approximately 0.25m away from each of the channel stations. An additional small boat was assigned to each grid to sequentially sample the vertical velocity profile at each station throughout the tidal cycle. These supplementary measurements at Station A were obtained using a Bendix Q-9 Savonius rotor current meter while those at Station B employed a CM2 propeller current meter. Both units were equipped with deck read-out modules.

b. Analytical Methods

Each drawn water sample was stored in prewashed glass jars and returned to the laboratory where the following analyses were conducted:

1. Sample Salinity

The salinity of each drawn water sample was determined using an induction salinometer (Beckman Model RS7-B). Measured conductivity was converted to salinity by comparison to Copenhagen Standard Sea Water.

2. Suspended Material Concentrations

Approximately one liter of each sample was vacuum filtered within forty eight hours of acquisition using dried and preweighed Nucleopore filters (47mm dia.-0.45 μ pore size). Filters were mounted in standard Millipore vacuum filtration setups. Following a thorough wash to remove salts, each filter was dried and reweighed to determine the by-weight concentration of the total suspended particulate matter present in the sample.

3. Grain Size Distributions

The particle size distributions characteristic of the suspended materials present in selected samples were determined using a Coulter Counter Model TA. A single aperture tube of 100 μ m diameter was used for all measurements.

4. Organic Carbon Content

Samples for the determination of particulate and dissolved organic carbon content were prepared by vacuum filtering fifty to 150 mls (varying as a function of the suspended material concentrations) of each sample using a Reeve Angel glass fiber filter (#934AH; 2.4 cm-0.45 μ). Each filter had been precombusted at 400°C for a period of two hours to remove volatile organics. Following the completion of filtration, filters were placed in small aluminum pans and frozen. Three 5 ml aliquots of the filtrate were pipetted into 10 ml glass ampoules which had been pre-combusted at 450°C for four hours. The ampoules were individually covered with a small piece of aluminum foil and frozen.

Particulate and dissolved organic carbon were determined by the conversion of organic matter to CO₂ through wet combustion with potassium persulfate, similar to the method developed by Menzel and Vaccaro (1964). The following additions were made to the ampoules containing filtered sample water. 1) 0.2g potassium persulfate (K₂S₂O₈) and 6% H₃PO₄ were introduced into the ampoules. 2) Oxygen gas which had been purified by passing

through a catalyst tube containing heated (400°C) cupric oxide was bubbled through for 5 mins to remove inorganic carbon components. Ampoules were sealed with a microburner. After sealing, the samples were oxidized in a standard laboratory autoclave set at 130°C for a period of four hours. The tops of the ampoules were then broken under sealed conditions in a nitrogen atmosphere and the $\text{CO}_2(\text{g})$ produced by the persulfate oxidation was carried by a stream of nitrogen gas to an IR analyzer and recorder. The peak heights produced by the CO_2 were corrected for reagent blanks and compared to a standard curve which had been determined by the wet oxidation of known amounts of dextrose ($\text{C}_6\text{H}_{12}\text{O}_6$).

To determine particulate organic carbon, the frozen filters were rolled up and placed in precombusted ampoules to which 5 mls of distilled water were then added. From this point the samples were treated in the same manner as the filtrate.

Results and Conclusions

Water Temperature

Water temperatures in the lower Thames River during the 1974-1976 study period displayed a seasonal cycle (Figs. 2 & 3) characteristic of northeastern United States coastal waters. Maxima at all stations occurred during July or August with minima observed during January, February, or March. Only slight variations were observed in this cycle. Water tempera-

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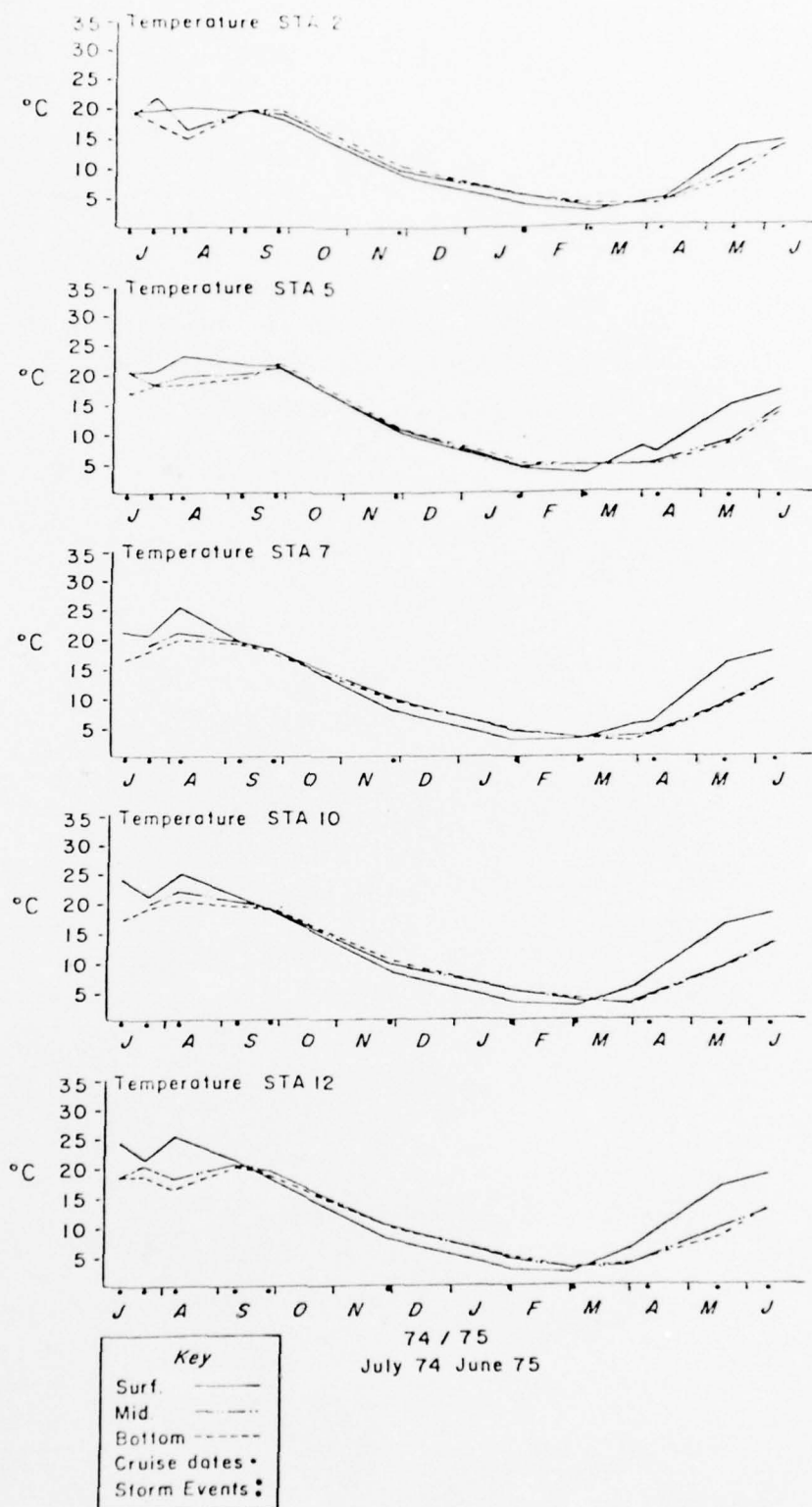


Fig. 2. Thames River Temperature Distributions 1974-1975

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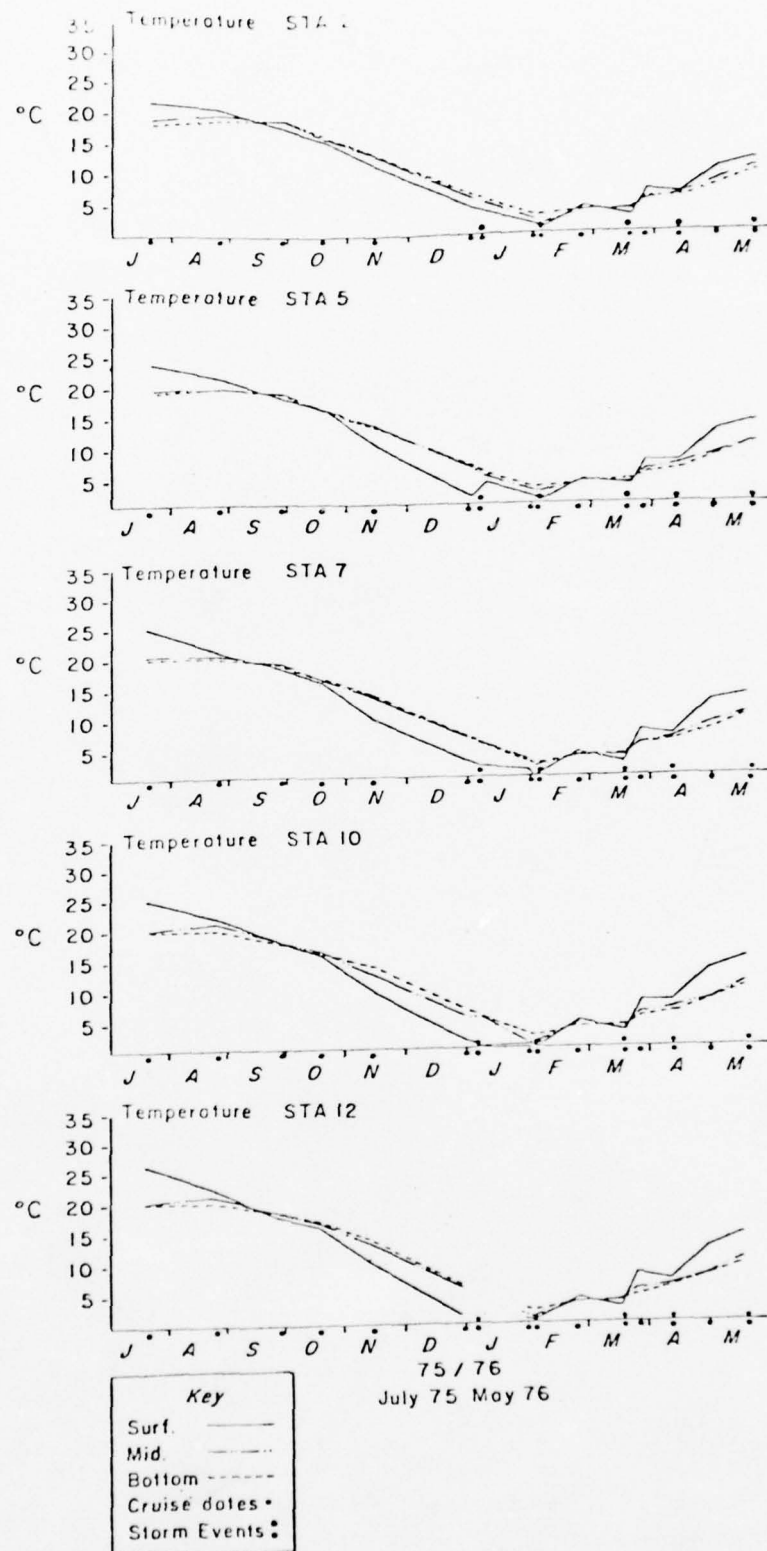


Fig. 3. Thames River Temperature Distributions 1975-1976

tures during the winter months of 1974-75 were slightly higher than those of 1975-76 favoring the persistence of near isothermal conditions for more than two months. As a result, the character of the spring increase observed during each survey year varied significantly. In 1974-75, initial warming in the surface waters was observed in March with increases at mid-depth and near bottom being delayed until April. In 1975-76 the spring increase occurred uniformly during late January and early February.

Water temperatures within the study area generally displayed minor spatial variability. In all cases, cross-stream (east-west) thermal differences were negligible. Analysis of the long-stream (north-south) variability shows a slight, but measurable, increase in annual temperature range as a function of distance from the mouth of the river. In the vicinity of Southwest Ledge, Station 2 (Fig. 1), the temperature range over each survey year averaged approximately 20°C. Proceeding to the north, the annual range progressively increases, finally exceeding 25°C at Station 12, adjacent to the Submarine Base.

Temperature distributions over the vertical display a regular seasonal cycle consistent with the yearly trends in air temperature. During spring warming and through the summer months, surface temperatures exceed those at mid-depth and near bottom. With the onset of cooling in September, this gradient reverses and surface temperatures fall below those observed at depth. This condition persists through the winter

months.

The fact that both positive and negative vertical temperature gradients are regularly present indicates that water temperatures represent a relatively minor influence on the density field within the lower river. In common with most estuaries, density appears to be primarily a function of salinity while the temperature structure and, in particular, the presence or absence of a thermocline, is at best a useful indicator of the intensity of vertical mixing. For example, during most of the year near isothermal conditions are indicative of free vertical exchange and limited density stratification. Conversely, significant thermal gradients most often indicate a reduction in vertical mixing and an increase in the degree of stratification in the salinity field. Application of these criteria in the examination of the Thames River thermal structure (Figs. 2&3) indicates a general decrease in vertical mixing with distance upstream from the mouth. The stations adjacent to Long Island Sound display nearly isothermal vertical profiles throughout the year while stations near the northern limit of the study area are characterized by persistent vertical gradients. In addition, the temperature distributions indicate that the degree of vertical stratification in the density field will increase progressively upstream. Given the character of the density field noted above, increased stratification can only be the result of developing vertical gradients in the salinity

field, a relationship to be examined in more detail in the next section.

Salinity

Salinity distributions at all stations display an irregular seasonal variability (Figs. 4&5). With few exceptions, mid-depth and near bottom values were essentially constant with salinities remaining nearly equal to those observed in adjacent Long Island Sound. Surface salinities were measurably lower than those observed at depth and displayed an erratic but evident seasonal variability roughly correlated with streamflow. Maximum salinities were observed in July or August with minima typically found in late winter or early spring.

The marked difference between surface and at-depth salinity produces a salinity field characterized by a persistent vertical gradient and a moderate degree of spatial and temporal variability. The nearly isohaline character of the deep waters causes the intensity of the vertical gradients to vary temporally as a function of surface salinity which, in turn, is approximately dependent on streamflow. Spatially, gradients vary as a function of distance from the mouth of the river with intensity progressively increasing along a northerly transect, a feature previously inferred from the water temperature profiles. East-west variability is for the most part negligible.

Analysis of the magnitude of the vertical salinity gradients and comparisons with concurrent water temperature gradients

A-15

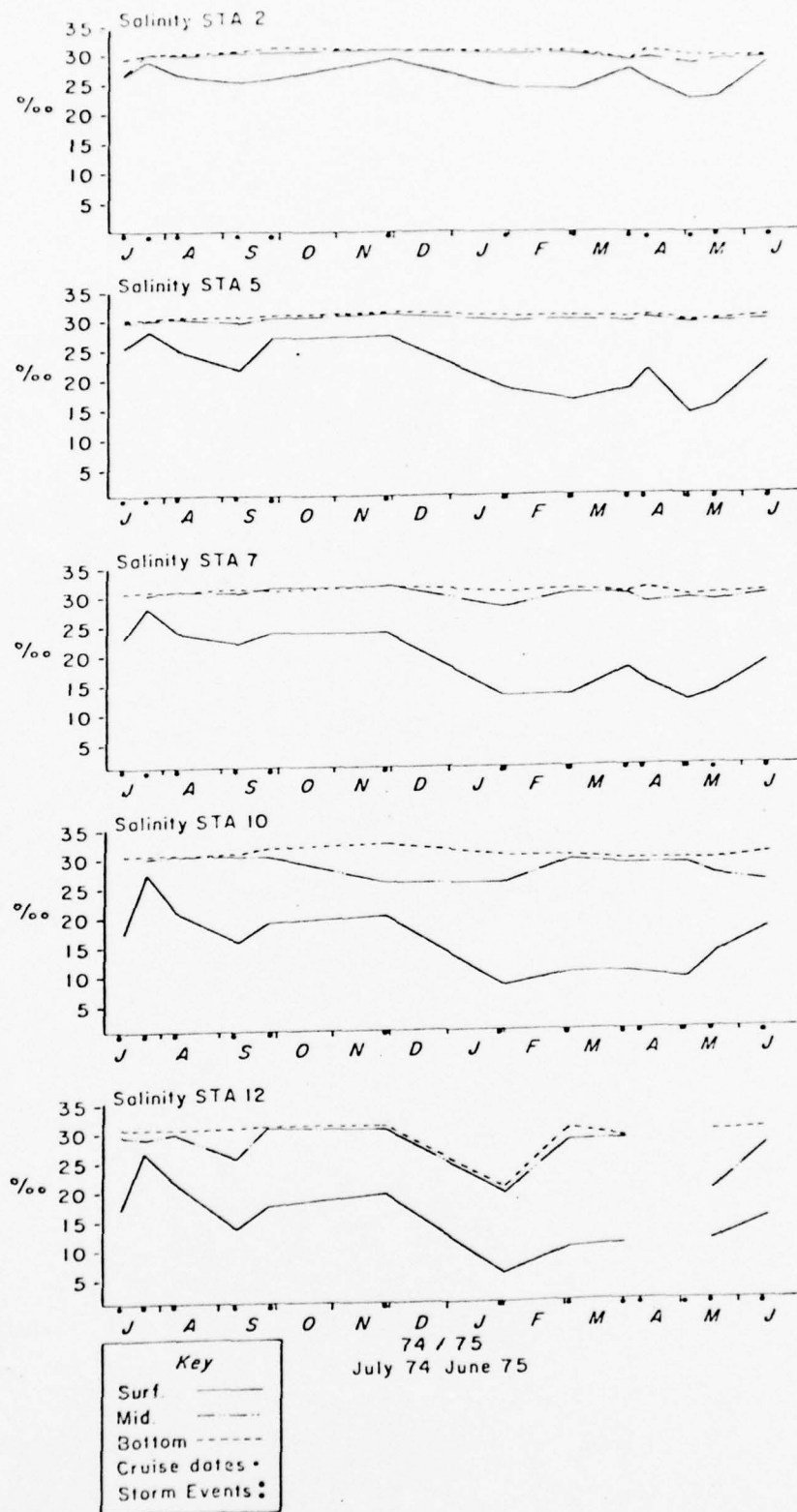


Fig. 4. Thames River Salinity Distributions 1974-1975

A-16

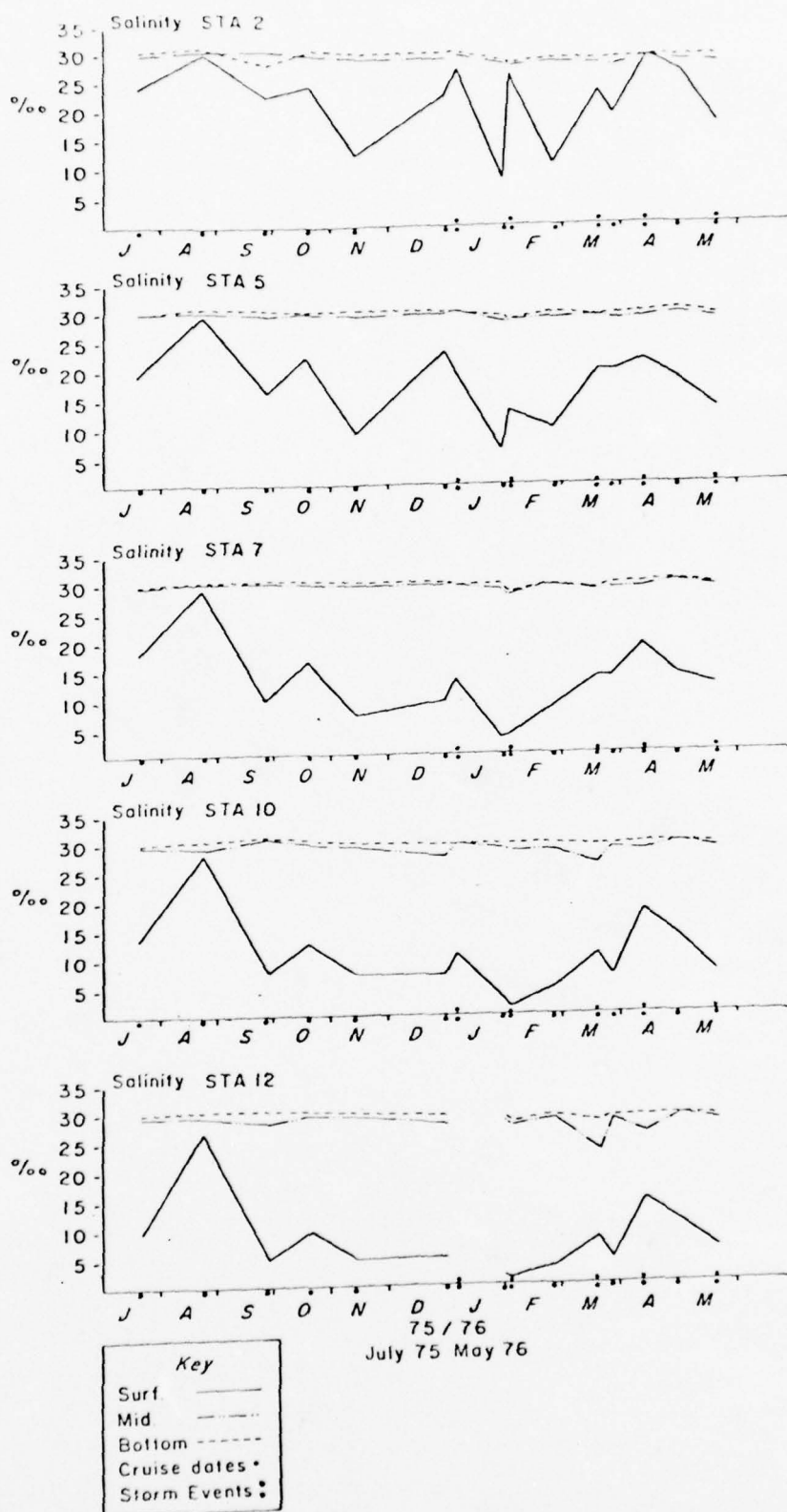


Fig. 5. Thames River Salinity Distributions 1975-1976

indicate that salinity clearly dominates the density field within the lower Thames River. Salinity gradients range from a low of $2^{\circ}/\text{oo}$ to a maximum of $26^{\circ}/\text{oo}$. Concurrent water temperature gradients seldom exceed 5°C and more typically average 2°C . Since a $1^{\circ}/\text{oo}$ change in salinity produces a variation in density equivalent to a 4°C change in water temperature it is apparent that salinity dominates and produces a density field with a high degree of vertical stratification. This density stratification is particularly pronounced in the near surface waters. Measurements obtained under a variety of streamflow conditions (Figs. 6&7) show maximum temperature and salinity gradients occurring approximately 2 to 3m below the surface. The near isohaline conditions above and below the transition zone indicates that the density field within the lower Thames can be adequately described as a two layer system with limited vertical mixing. Moreover the persistence of mixing limitations despite the presence of moderately energetic meteorological events suggests that this area may be effectively sheltered from storm events. Each of these characteristics are of particular importance within considerations of suspended material transport in and through the estuary.

Suspended Material Characteristics

a. Concentrations

Suspended material concentrations observed in the lower Thames River during the years 1974 through 1976 (Figs. 8&9)

22 July '74

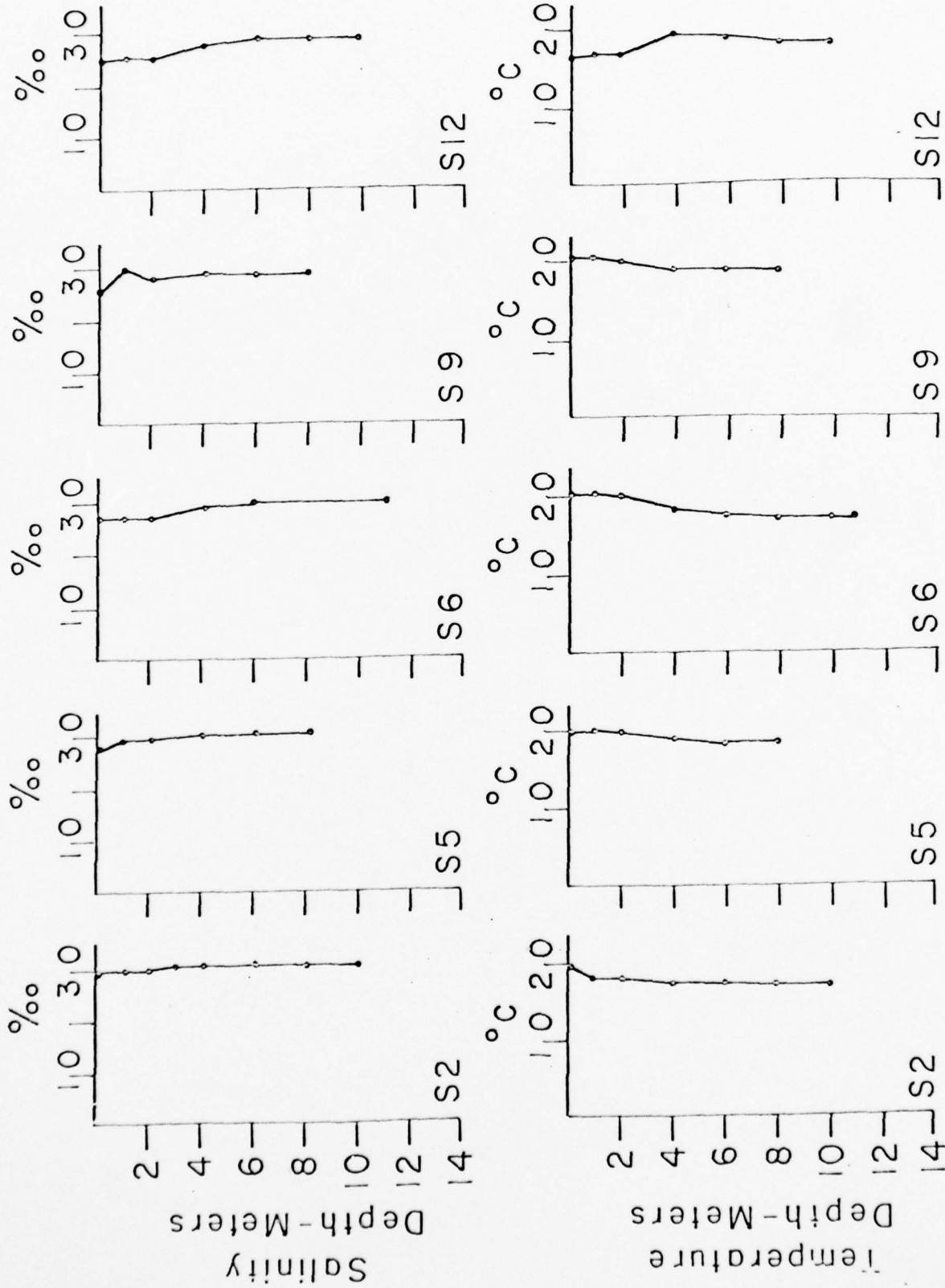


Fig. 6. Thames River: Vertical Temperature and Salinity Structure July 1974

1 April '75

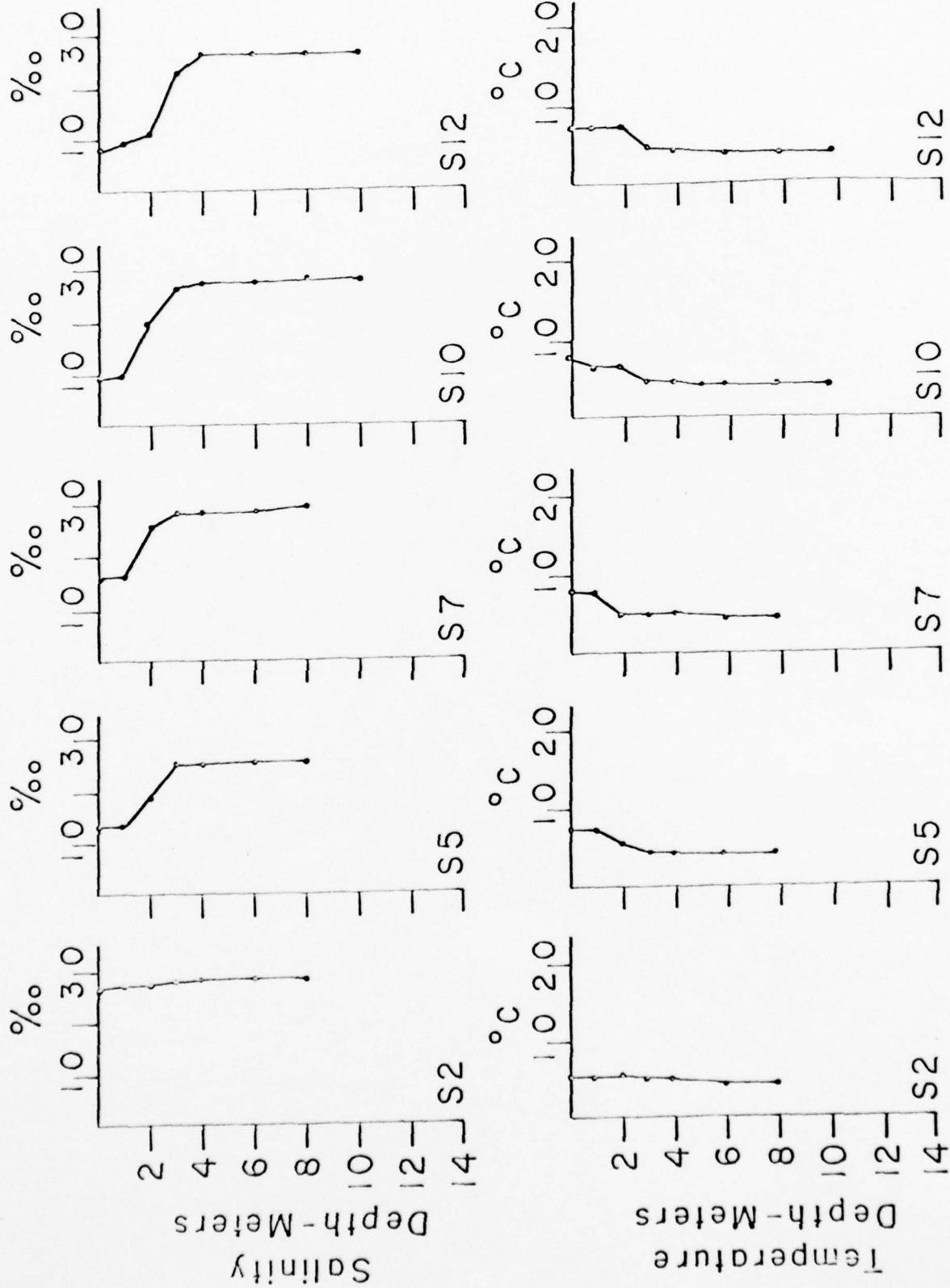


Fig. 7. Thames River: Vertical Temperature and Salinity Structure April 1975

A-20

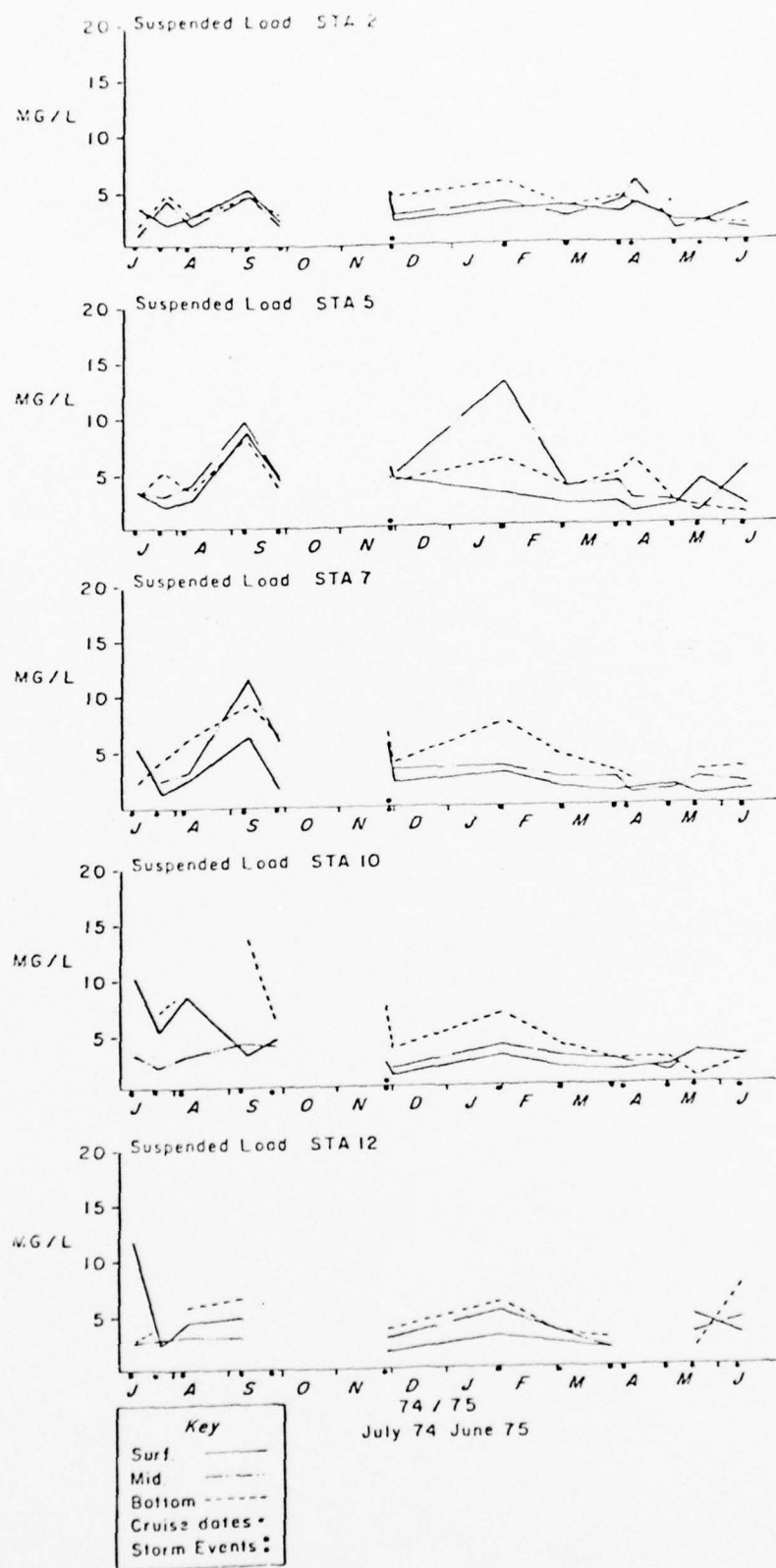


Fig. 8. Thames River Suspended Material Concentrations 1974-1975

A-21

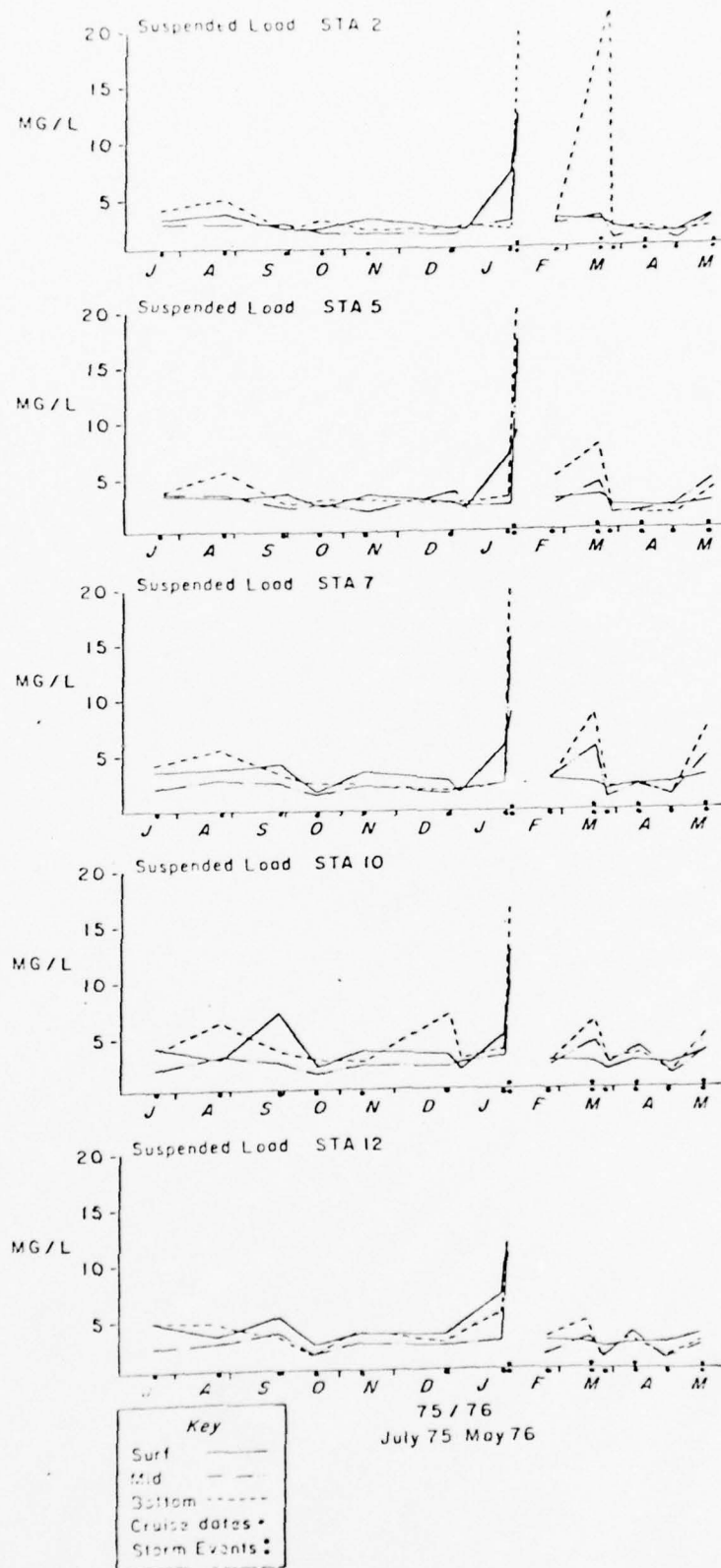


Fig. 9. Thames River Suspended Material Concentrations 1975-1976

displayed no evident seasonal variability. Although contrasting sharply with previous observations in other estuaries (Postma, 1967; Meade, 1968; and Schubel, 1969) this behavior is consistent with recent observations obtained in eastern Long Island Sound (Bohlen, 1975) and seems thoroughly representative of a sediment-poor system in which transport and ultimately the concentration of suspended materials is dominated by a variety of factors including stream flow, tidal state, and wind stress. The relative importance of each of these factors varies both spatially and temporally. Since these cycles vary independently and seldom display simple phase relationships, the resultant suspended material field is often characterized by erratic variability and an absence of obvious periodicity. Such behavior is particularly common in systems characterized by low ($<5\text{mg/l}$) suspended material concentrations.

Examination of the survey data reveals a relatively minor degree of spatial variability in the suspended material field. During 1974-75 (Fig. 8) upstream concentrations tended to be slightly higher than those observed near the mouth of the river and in adjacent Long Island Sound. Maximum concentrations were highest in the vicinity of Stations 5 and 7. In this same period concentrations over the vertical generally increased progressively with depth. Surface concentrations were regularly less than those observed at mid-depth or near bottom.

The spatial characteristics of the suspended material field during 1975-76 were substantially different than those of

1974-75. Average concentration levels were essentially constant throughout the study area and displayed no tendency to increase in the upstream direction. Over the vertical, concentration gradients varied irregularly with surface and near bottom concentrations alternately dominant. The sense of the gradients was generally correlated with streamflow. During periods of low discharge, concentrations at depth exceeded those observed at the surface. Under these conditions the concentration field appears to be primarily dependent on shear stress distributions at the sediment-water interface. Increasing discharge modifies this dependence and eventually causes a reversal in the sense of the concentration gradient. Streamflow begins to dominate and surface concentrations typically exceed those observed at depth. The relatively infrequent occurrence of gradients dominated by surface values indicates that streamflow generally represents a second order influence on the suspended material field within the lower Thames River. A comparison of volume discharge to concurrent intertidal volume further supports this construct.

b. Organic Composition

To examine the relative concentrations of organic material in suspension within the study area (Fig. 1) the particulate organic carbon content of each drawn water sample was determined. These data are of particular importance in the analysis of the biological impact of the dredging operations and were also intended to supplement concurrent work on filter feeding organisms (See section B). As indicated above, the analyses employed wet oxidation techniques. These methods have in the past been criticized for failing to provide accurate determinations of total organic carbon. Work by Sharp (1973), Stanikova (1970), Skopintsev, *et al.*, (1966) and Skopintsev (1968), analyzing low oceanic concentrations ($<2\text{mg/l}$) by direct combustion methods have produced carbon concentrations differing from those obtained using wet oxidation techniques (e.g. Menzel and Vaccaro, 1964) by more than a factor of two. The cause of these differences appears to be the structure of the organic carbon compounds associated with each sample. Earlier investigations of the efficiency of the wet oxidation techniques indicated that although the procedure provided very nearly 100% recovery of compounds such as sugars, amino acids and fatty acids, its efficiency decreased significantly if high concentrations of refractory long chain or polycyclic hydrocarbons were present (Menzel and Vaccaro, 1964; Fredericks and Hood, 1965; Strickland and Passons, 1968; Williams, 1969).

These results suggest that the observed differences between the wet oxidation and direct combustion techniques are caused by significant concentrations of refractory compounds rather than the general inefficiency of the wet oxidation procedures. These refractory compounds are relatively stable and generally considered resistant to biological breakdown. As such they represent a minor source of nutrients and for the purposes of biological assessments can generally be neglected. Under these conditions particulate carbon concentrations are accurately assayed using wet oxidation techniques.

Particulate organic carbon concentrations observed in the lower Thames River during 1974 through 1976 are shown in Figures 10 and 11. The distributions display a weak seasonal variability with maxima observed in October and March. The increased concentrations during these periods must be considered to be the result of autumnal and late winter plankton blooms respectively, although each peak is significantly delayed with respect to the normal bloom period (Riley, 1959). Minima typically occur during December-January. The marked increase observed during February, 1976 was the result of a major storm event and will be discussed in more detail below.

The spatial variability of the carbon distributions is extremely limited. Horizontally, concentrations are nearly constant and no measurable station-to-station trend can be discerned. Over the vertical, concentrations tend to decrease

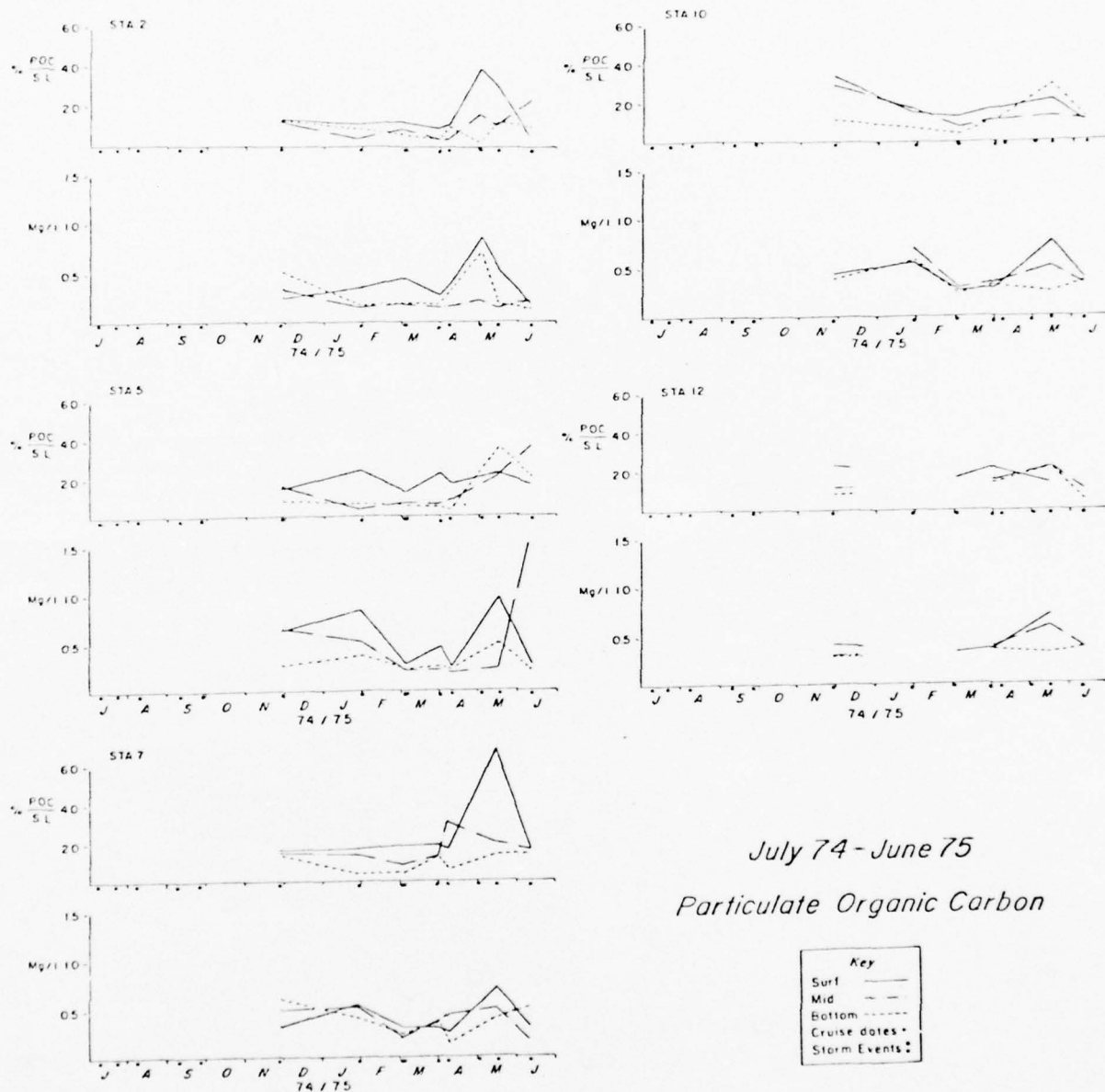


Fig. 10. Thames River
Organic Carbon Content 1974-1975

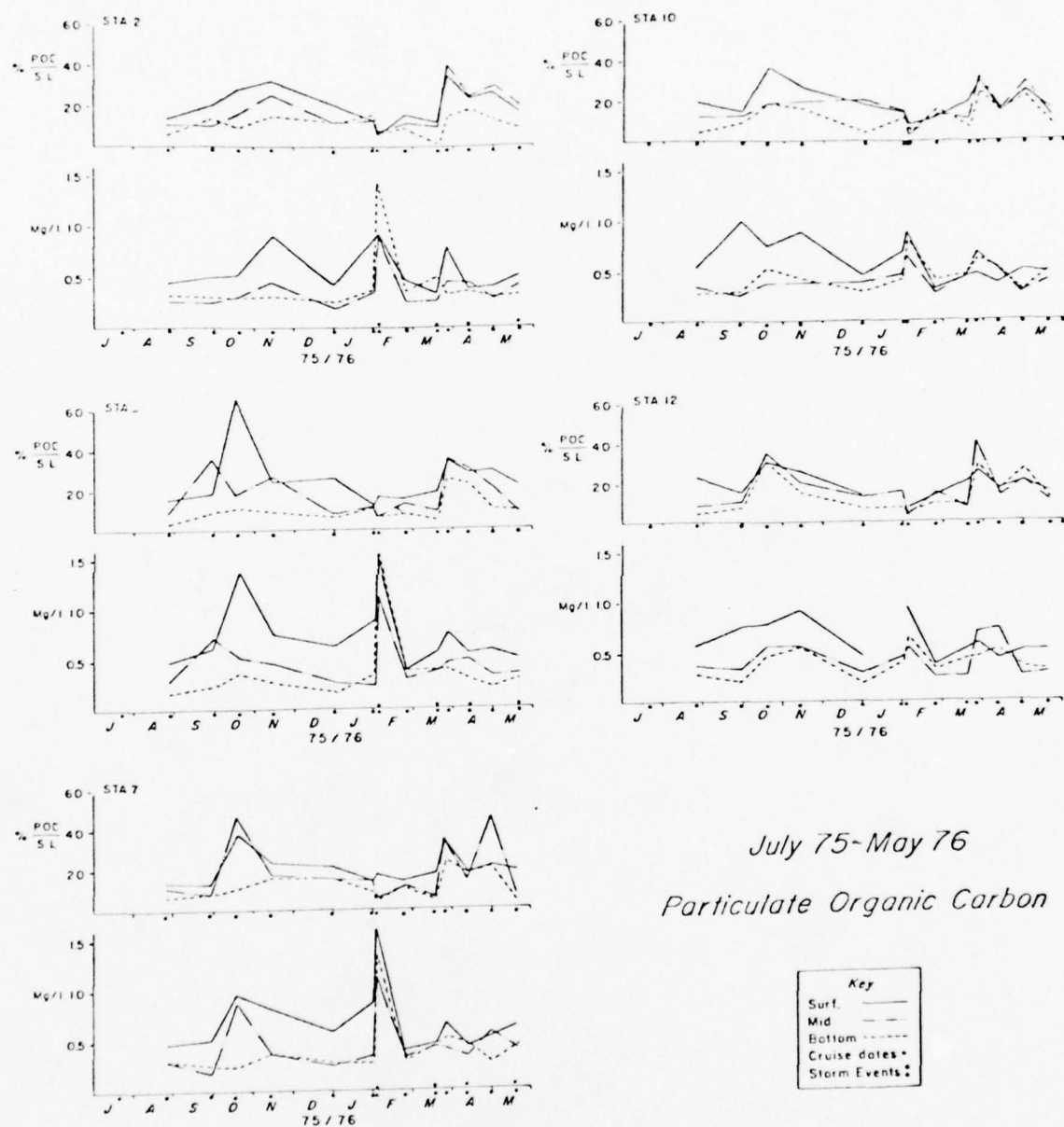


Fig. 11. Thames River
Organic Carbon Content 1975-1976

with depth and surface values often exceed those near bottom by a factor of two. Vertical gradients display no reasonable correlation with streamflow and rather appear to be more sensitive to biological factors including nutrient flux and ambient light than hydrographic conditions.

Particulate organic carbon typically constitutes 20 to 30% of the total suspended load in the lower river (Figs. 10-11). These levels, implying that up to 40 to 60% of the suspended materials are organic in origin, are consistent with previous observations in Long Island Sound (Meade, 1972; Bohlen, 1975). The percentage of organic materials in suspension displays a seasonal variability similar to that observed in the particulate carbon distributions. This simple variability is the result of the generally low concentration levels and limited seasonal variability characteristic of the suspended material field. These background characteristics simplify resolution of dredging or storm induced alterations in the composition of the suspended load. In both cases particulate organic carbon concentrations will increase significantly, often by a factor of three or more. Total suspended material concentrations, however, typically increase by a factor of ten. As a result the relative percentage of organics decreases significantly during these anomalous events. The storm observed in February, 1976 illustrates this variability (Fig. 11). The significance of these phenomena will be discussed in more detail below. For the present, it is

simply useful to note that the behavioral characteristics of the suspended material field in the Thames River permits ready discrimination between average and aperiodic transport events.

Transport Characteristics

The density field characteristic of the lower Thames River favors a persistent two layer flow regime in which southerly or downstream average mass transport prevails in the upper, near surface layer while northerly or upstream transport persists at depth. Observations indicate that the thickness of the surface layer varies between 2 to 3 meters in the area south of Stations 9 and 10 (Fig. 1). North of this section to the Submarine Base, layer thickness increases slightly and becomes more variable and sensitive to streamflow and tidal state. Within this surface layer measured velocities display a regular variability dominated by the semi-diurnal tides and freshwater discharge. During the surveys conducted as part of this study, this combination produced maximum velocities of approximately 50 cm/sec. Below the surface layer a relatively low energy circulation system prevails with velocities seldom exceeding 25 to 30 cm/sec. These velocities display a regular, tidally dominated, variability and appear to be relatively insensitive to streamflow.

The persistence of pronounced stratification within the lower Thames is a relatively unique feature in an estuary

characterized by streamflow values (Fig. 12) small in comparison to the intertidal volume. Typically, such systems display gradual variations in density over the vertical with the degree of stratification varying seasonally as a function of streamflow. The characteristics observed in the Thames indicate that the study area is subject to a relatively low degree of vertical mixing. As a result the freshwater outflows are effectively ejected as a well defined, coherent layer over the higher density coastal waters. These conditions are apparently the result of the orientation and morphology of the river basin. The narrow channel, high banks and north-south alignment of the river effectively reduce and in many cases eliminate the influence of wind induced mixing. Review of the wind field characteristic of the study area (Figs. 13 and 14) indicates that maximum energies are typically confined to the east-west components and suggests that of the above factors, channel orientation may represent the principal determinant limiting wind effects.

The absence of significant wind induced turbulence combined with the regional geological characteristics results in a relatively simple sediment transport system in the lower Thames. In shallow estuaries limited turbulent mixing affects not only the density field but also the character and magnitude of boundary shear stress acting on the sediment-water interface. In the absence of this factor interfacial shear and ultimately the competence of the flow field to erode and/or transport sediments is often measurably reduced. Concurrently, transport variability

Fig. 12.

THAMES RIVER

ESTIMATED MONTHLY AVERAGE STREAMFLOW
ENTERING LONG ISLAND SOUND

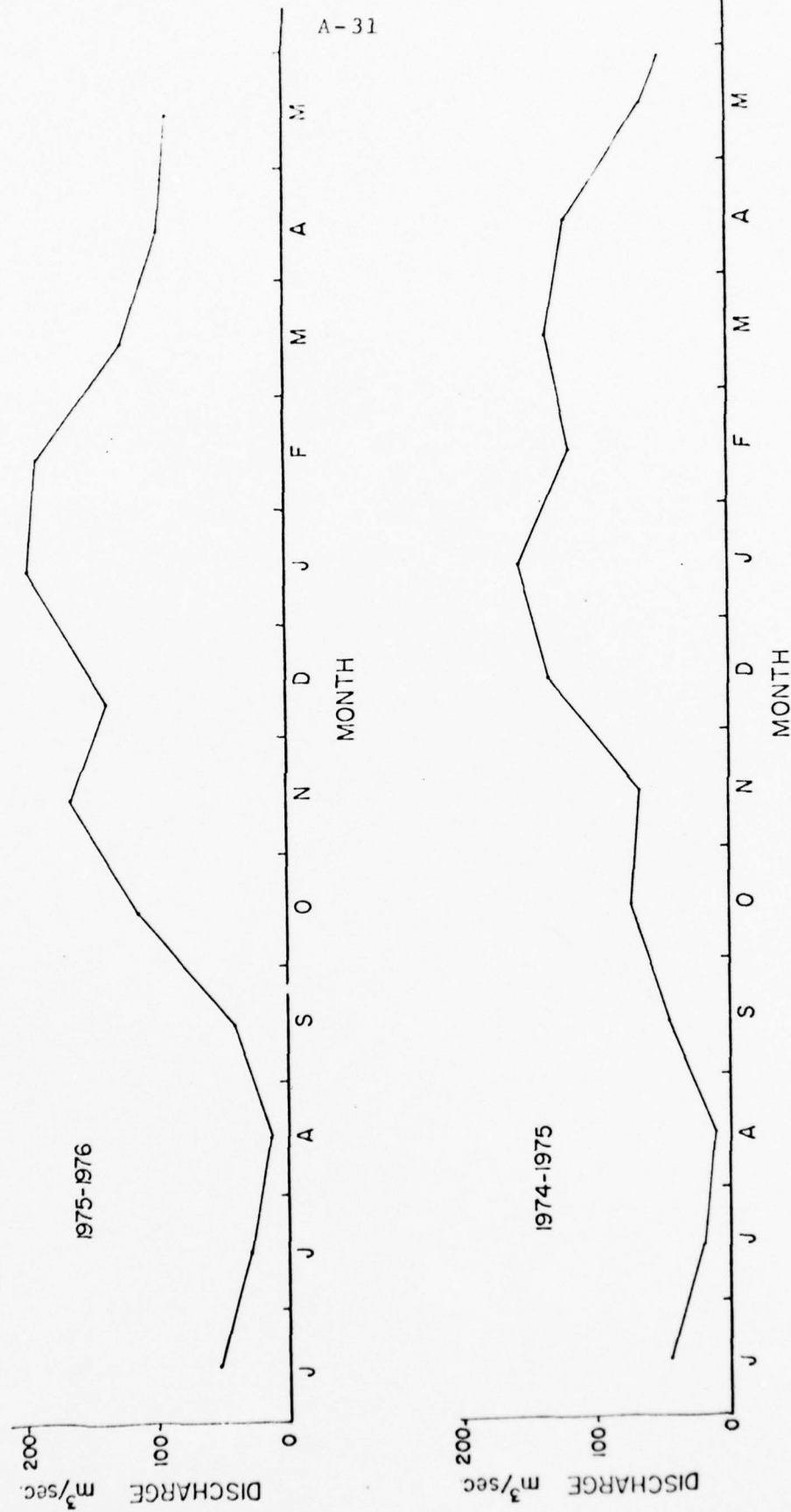


Fig. 13.
DAILY AVERAGE WIND VELOCITY

MILLSTONE POINT 10m level

JULY 1974 - JUNE 1975

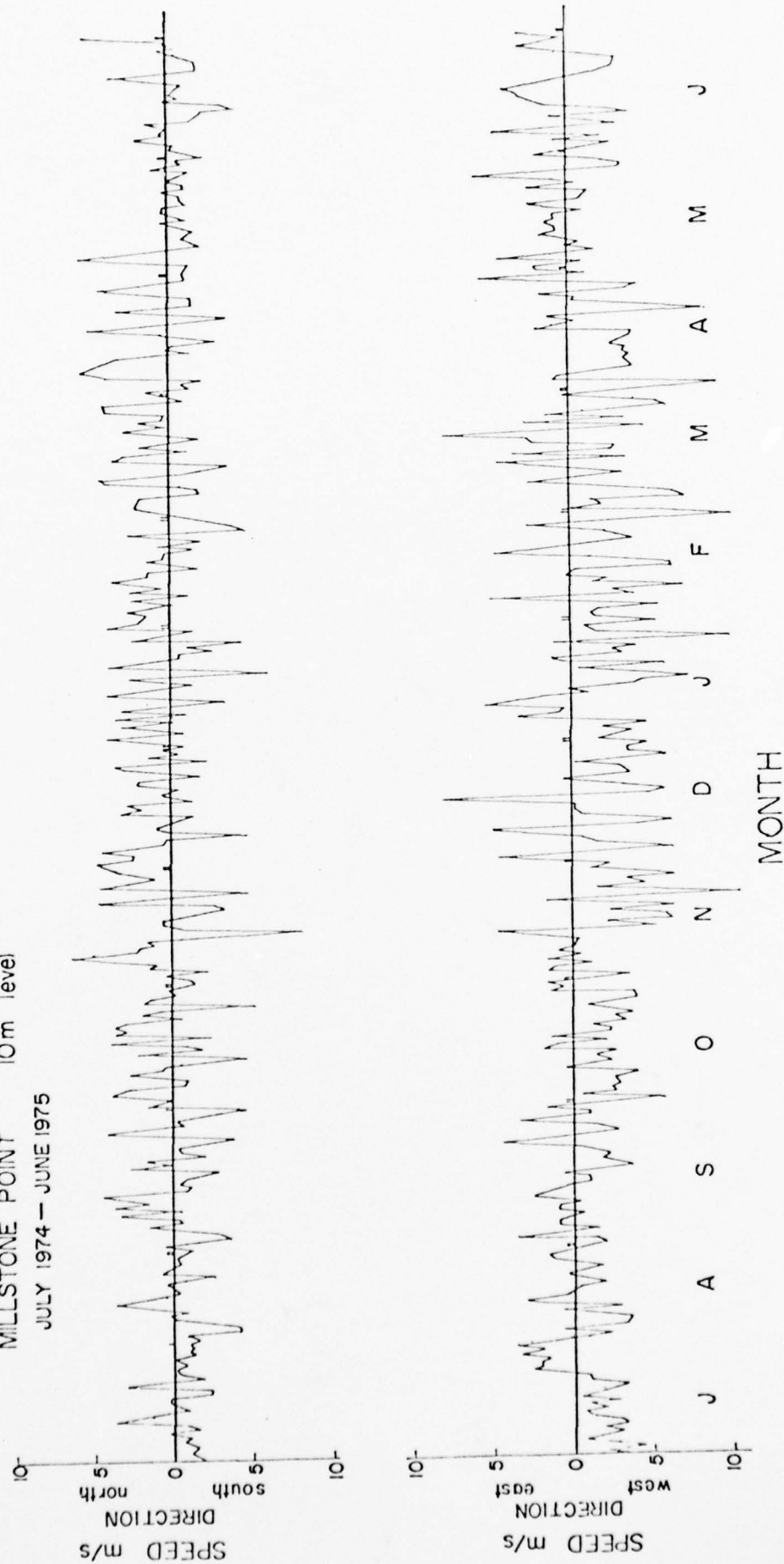
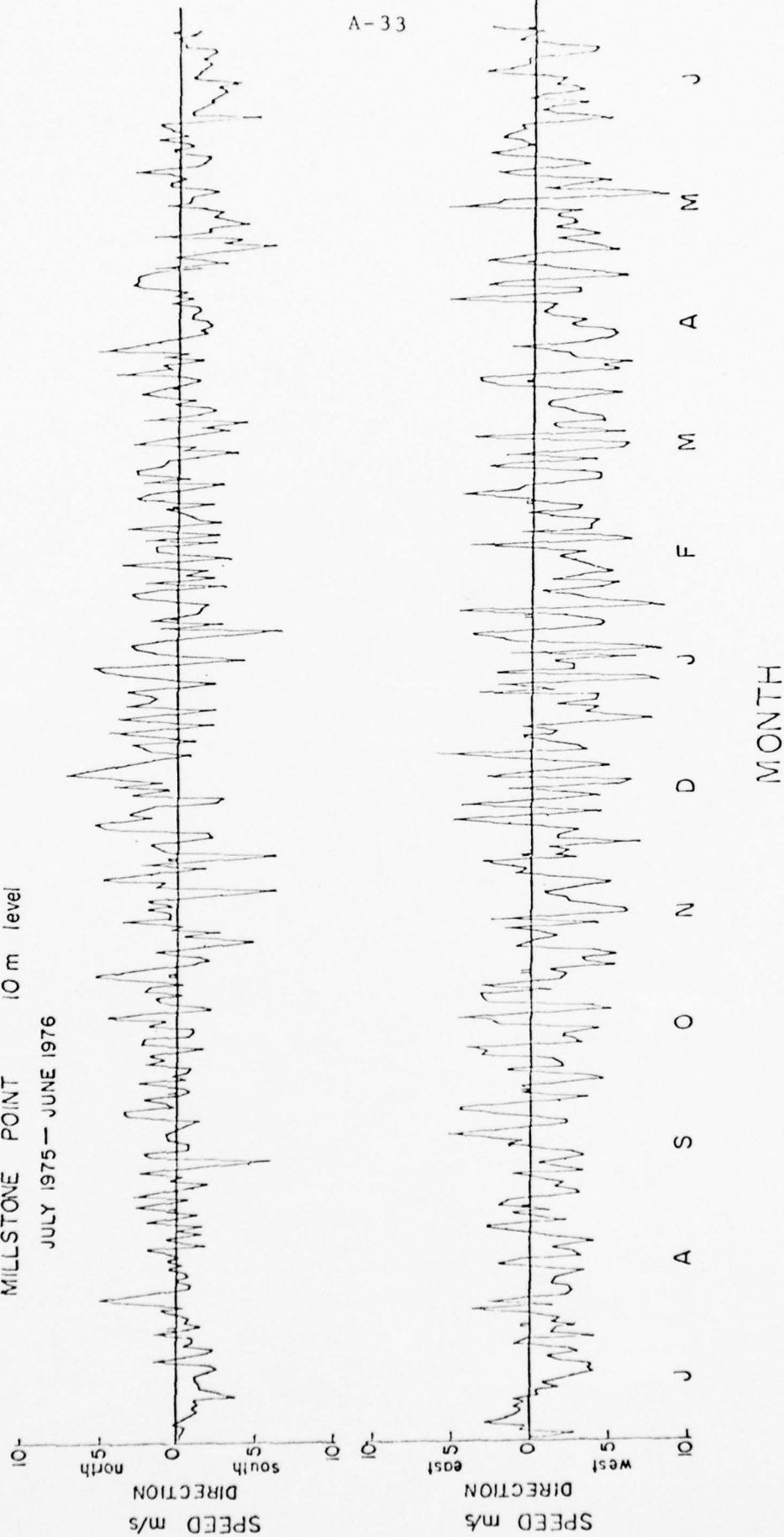


Fig. 14.
DAILY AVERAGE WIND VELOCITY

MILLSTONE POINT 10 m level
JULY 1975 - JUNE 1976



is reduced and material flux becomes more nearly a simple function of the tidally dominated velocity field. In such a system the qualitative characteristics and selected quantitative features of the sediment transport system can be deduced by considering the magnitude of the peak velocity, details of the residual flow and the mechanical/geological properties of the sediment bed.

Evaluation of the above features within the Thames River provides clear indications that this system is best characterized as a low energy regime in which recent hydraulic and geological factors favor net accumulation of sediments. Near bottom velocities are consistently low and seldom exceed values necessary to erode the fine grained cohesive sediments found throughout the lower river. As previously mentioned upstream drift prevails at depth. Analyses of the current meter records indicates average flows of approximately 10 cm/sec. With negligible local resuspension, this drift, acting in combination with the factors controlling sediment transport in adjacent Fishers Island Sound, represents the principal determinant governing the volume of materials suspended and the fraction deposited in the estuary. Although this latter feature is difficult to evaluate without more detailed sedimentary analyses, preliminary estimates based on short term flow observations and assigned suspended material concentrations indicate that deposition rates in the lower Thames are low in comparison to those observed in other areas within and adjacent to Long Island Sound. Calculations

based on the velocity observations of May, 1975 and estimated average suspended material concentrations in the near bottom and surface layers yield an approximate deposition rate of 0.45 mm/year over the entire surface of the estuary. This rate is similar to that recently estimated for the central Long Island Sound system (0.33 mm/year; Bokuniewicz, et al., 1975) but differs by an order of magnitude from the most commonly referenced rate (4.5 mm/year; Thomson et al., 1975) again derived for the central Sound. Given the character of the deposition process and the many assumptions implicit in the calculations, immediate determination of the cause and significance of these differences cannot be provided. The low rates seem consistent with the sediment-poor character of the river basin and the dominance of offshore source areas. In addition, it should be realized that the computed values represent an average rate over the entire surface area of the estuary. Rates can be expected to display significant spatial variability, particularly in artificially deepened areas. Evaluations of historical data obtained at several sites similar in character to the Thames River indicate that deposition rates in dredged areas may typically approach 10 cm/year. Rates again, however, are highly variable and depend primarily on the extent and character of the dredging project.

Dredging Impacts

a. Near field

The most apparent (immediate) impact of the dredging operation on the suspended material field within the lower Thames was the direct introduction into the water column of high concentrations of fine grained materials due to leakage from the dredge bucket and overflow from the attendant barge (see Figure 1 of introduction). The results of the surveys, intended to determine the quantitative character and spatial extent of the perturbations produced by the operation, are shown in Figures 15, 16 and 17. The locations of each survey are noted on Figure 1. These observations indicate that in the worst case, the operating dredge and barge produced near field concentrations of total suspended solids in excess of 100 mg/l (Fig. 15). Given the character of the suspended material field in the lower river (Figs. 8 & 9), this represents a significant increase and exceeds average background concentrations by more than a factor of thirty. Calculations indicate that the mass of material contained in the turbid plume represents an increase in total suspended sediments in the estuary of approximately 10 to 20%. Despite this significant load, the surveys indicate that the influence of the operations is essentially limited to the immediate vicinity of the dredge and barge. In all cases concentrations reached maximum values within 100 yards of the barge. Proceeding downstream, values then rapidly decayed with

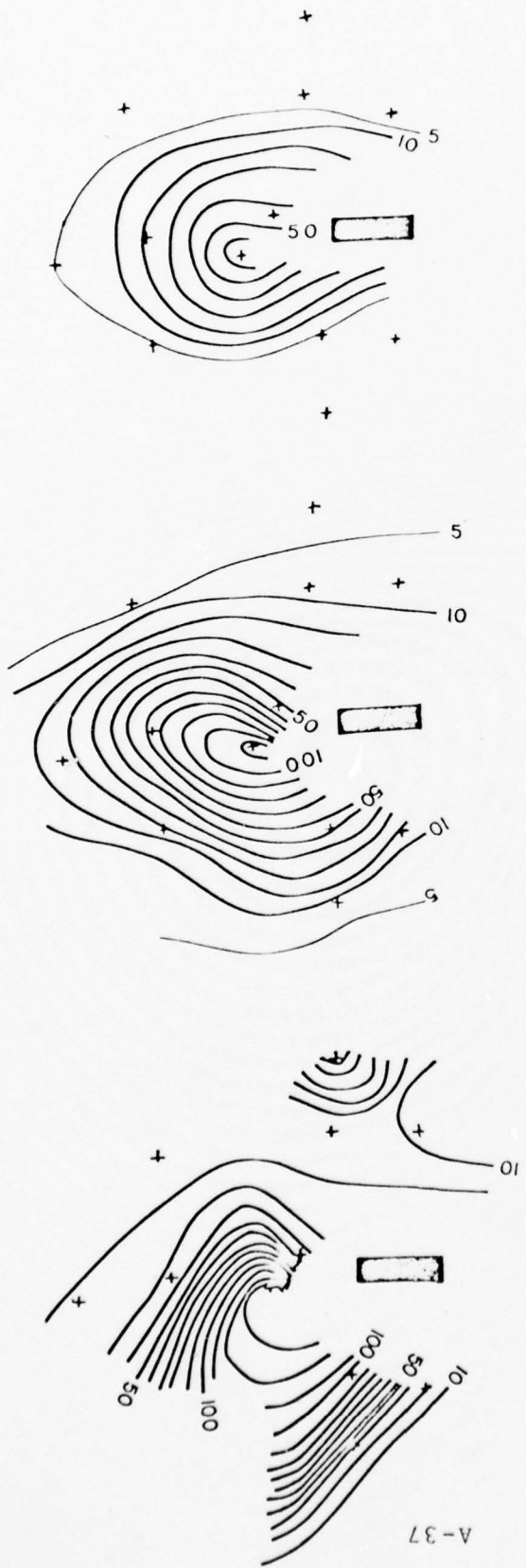
SURFACE

3 METERS

BOTTOM

100 YDS

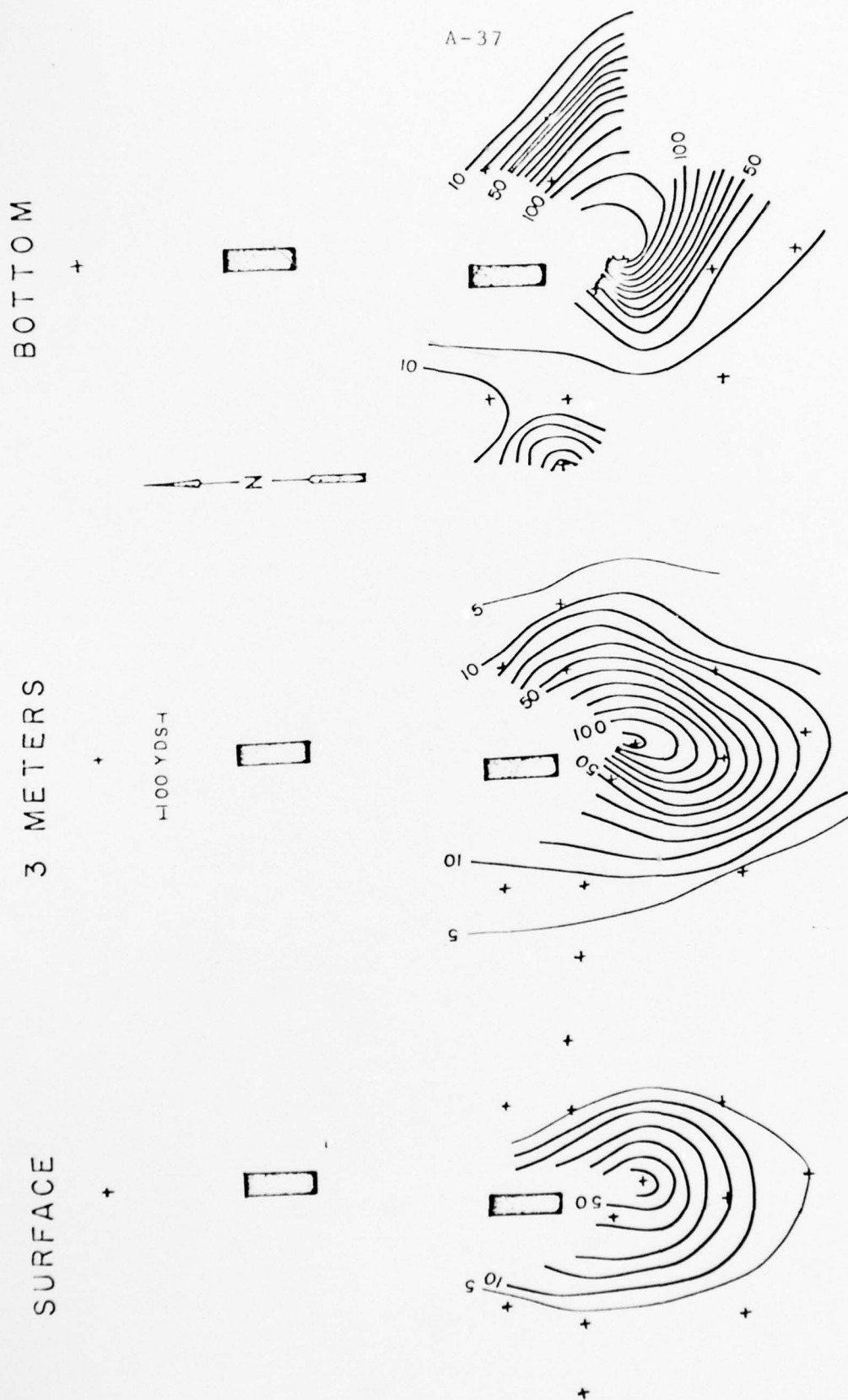
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MAP A

29 NOV 1974

Fig. 15. Thames River High Resolution Survey. Suspended material concentrations (mg/l) adjacent to operating dredge/barge at site A.

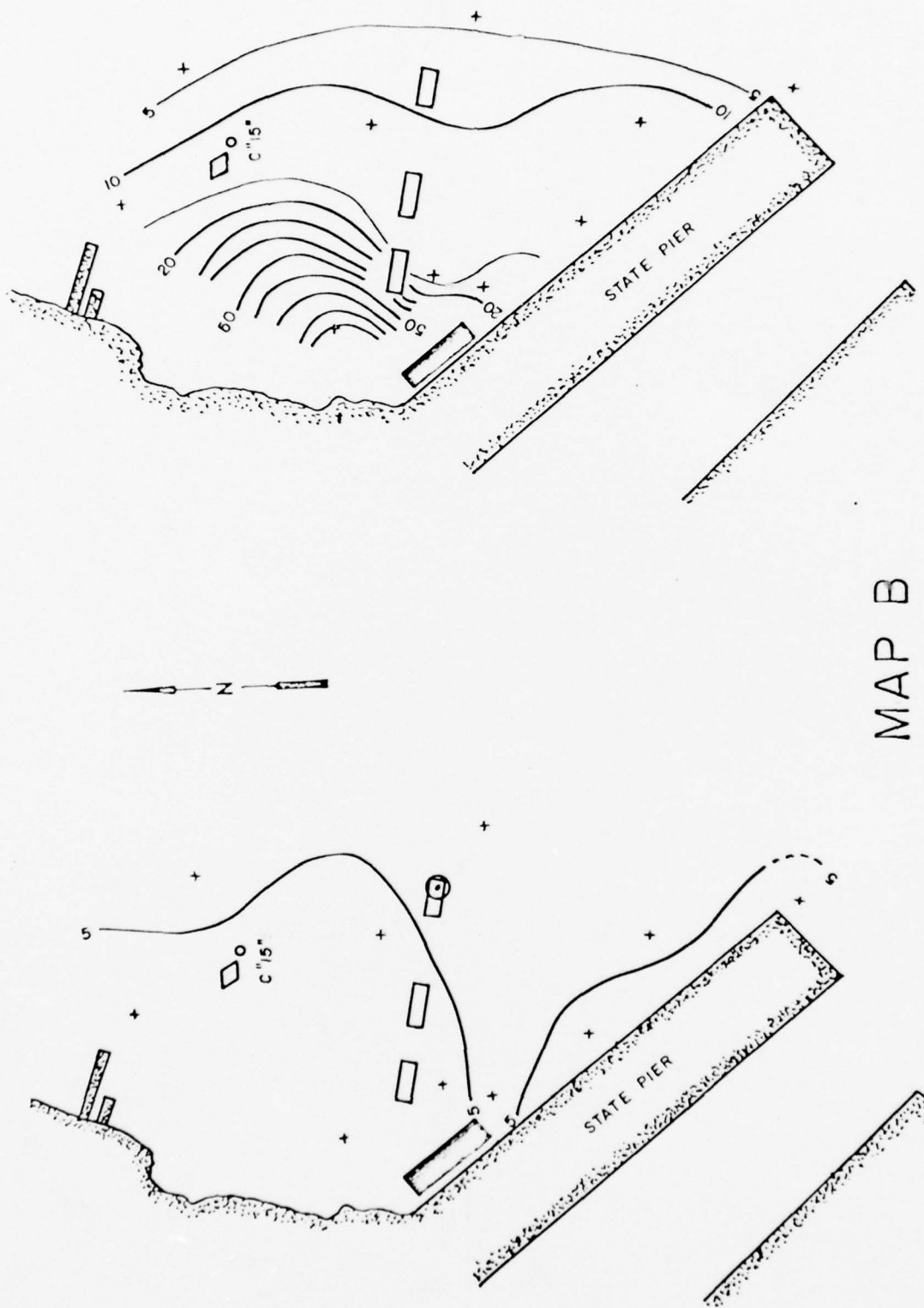


MAP A 29 NOV 1974

Fig. 15. Thames River High Resolution Survey. Suspended material concentrations (mg/l) adjacent to operating dredge/barge at site A.

SURFACE

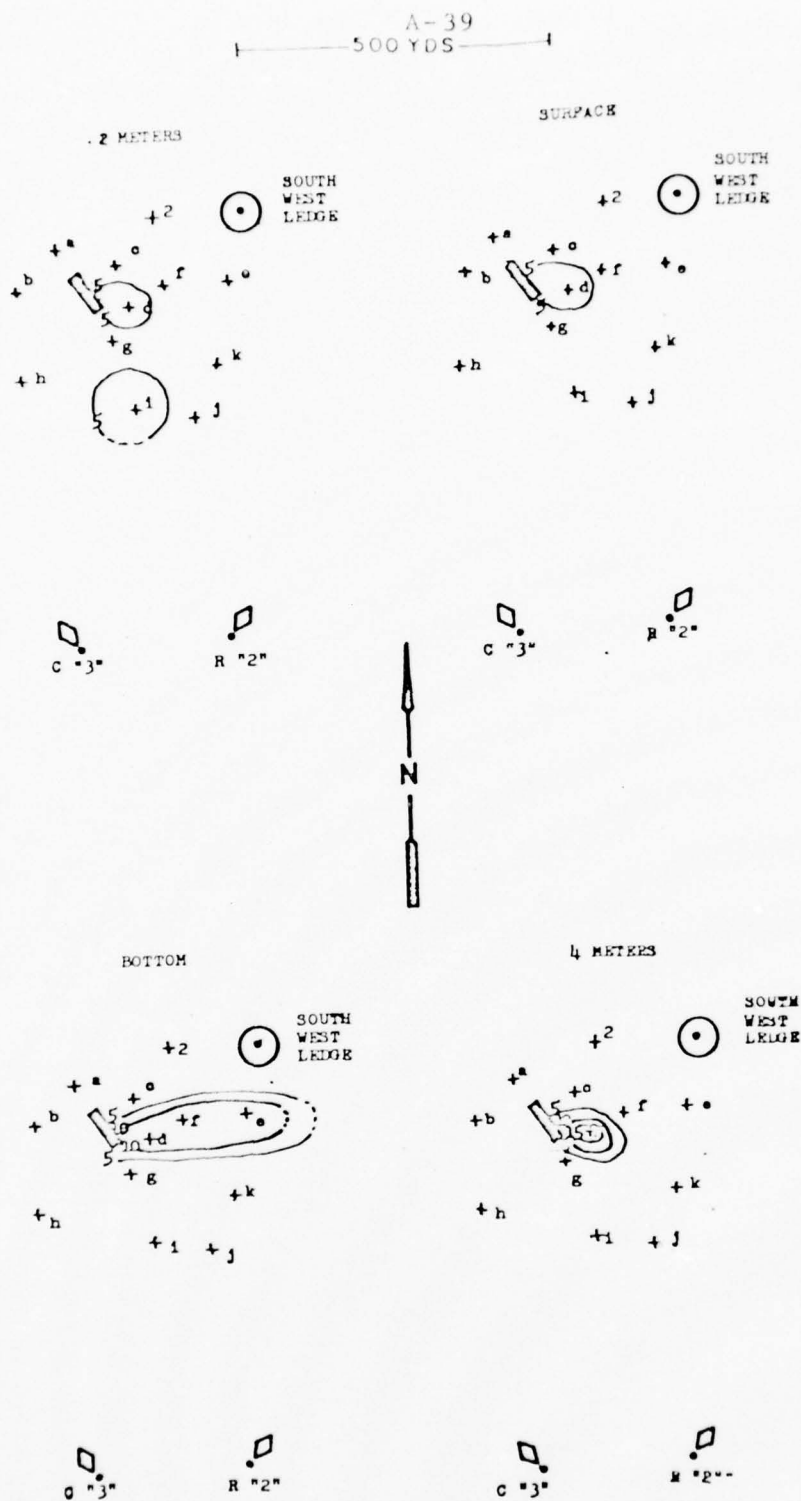
2 METERS



MAP B

14 MAY 1975

Fig. 16. Thames River High Resolution Survey. Suspended material concentrations (mg/L) adjacent to operating dredge/barge at site B.



MAP C

11 JUNE 1975

Fig. 17. Thames River High Resolution Survey
Suspended material concentrations (mg/l) adjacent to operating
dredge/barge at site C

concentrations returning to background within 300 to 500 yards of the dredge and barge. Initial reviews suggest that this pattern is primarily governed by the local flow field and is essentially insensitive to the characteristics of the sediment being dredged. The latter factor influences primarily material concentrations. Its effects on spatial distributions are obscured by the limited variability in sediment characteristics in the lower river.

In addition to the marked increase in local concentrations, the composition of the suspended load was measurably perturbed in the vicinity of the dredge and barge. Particulate organic carbon concentrations typically increased by a factor of two while the relative percentage of P.O.C. with respect to the total concentration of suspended solids decreased sharply (Fig. 10). Again, these variations are confined within the plume of turbid water and were not observed at stations which were distant relative to the dredge and barge.

b. Large scale effects

A major dredging project can be expected to modify the large scale characteristics of the suspended material field in an estuary by causing alterations in the composition of the sediment-water interface and through variations in the hydraulic regime and resultant transport induced by the increased cross-sectional area of the total river channel. The

first phase dredging in the lower Thames directly modified a portion of river bottom approximately 500 ft. wide and 3.9 miles long. Over most of this reach, project width represents less than 0.2 of the total river width. This fact, when considered in combination with the relatively cohesive character of the sediments and the low energy velocity field in the area, suggests that substrate induced alterations in material transport are most probably negligible. As previously discussed, channel deepening, rather than providing fresh, erodible bottom, generally exposes an area over which increased deposition rates will prevail. As a result, the sediment-water interface within the project area represents a minor determinant of the composition or concentration of the suspended material field.

Such a condition appears to prevail within the lower Thames. Direct evaluations of both suspended material composition and concentrations (Figs. 8-11) fail to provide any evidence of substantive alterations in the material field following completion of the dredging project. The characteristics observed during 1975-76 are essentially identical to those observed in 1974-75, and continue to display variability governed primarily by the conditions prevailing in adjacent Fishers Island Sound. Under these embayment conditions, local substrate variations within the lower Thames reasonably represent a negligible influence on the suspended material field.

A significant increase in depth and cross-sectional area

of an estuary can serve to measurably alter material transport by increasing the longitudinal extent of the intrusion of high salinity bottom water and by modifications in vertical mixing produced by variations in turbulent shear. In the absence of major alterations in streamflow or tidal characteristics, the magnitude of the transport variability can be qualitatively evaluated by considering the extent to which dredging has modified the average cross-sectional configuration of the estuary. Such evaluations applied to the lower Thames suggest that the phase one dredging project will result in negligible alterations in the volume or characteristics of suspended material transport.

As discussed above the phase one project deepened a relatively small portion of the total river width. Over this area, project specifications called for approximately a five foot increase in depth. At most sections this value represents a local increase in depth of nearly 15% but yields less than a 5% increase in the total cross-sectional area of the estuary. Review of previous efforts to predict dredging effects and the concurrent limits of saltwater intrusion (Harleman and Ippen, 1969) suggests that such minor variations will not produce a measurable alteration in salinity structure and associated transport. Examination of the salinity data obtained during 1974-1976 (Figs. 4 & 5) supports these conclusions and fails to provide any evidence of significant variations in vertical

structure or longitudinal distribution that can be simply related to dredging. The dredging induced alterations, if any, are slight and well below the resolution of the observational and analytical techniques employed in the survey. One can only conclude that large scale variations in material transport are of similar magnitude.

Discussion

Initial reviews of the survey data obtained during the period 1974 to 1976 indicate that the phase one or first increment dredging of the lower Thames River produced only small scale variations in the suspended material field. Measurable impacts were confined to the immediate vicinity of the operating dredge and barge and appeared to be the combined result of leakage from the dredging bucket and fluid overflow from the barge. In all cases the mass of materials introduced by the operation was estimated to be less than 20% of the total mass of sediments suspended in the estuary. This potential contribution, however, was effectively eliminated by the evidently high settling rates characteristic of the project related materials which favored simple near field perturbations.

When compared to suspended material variations associated with naturally occurring aperiodic storm events, the above dredging-related impacts appear negligible. During the storm of 1-2 February, 1976, for example, concentration levels throughout

the lower river increased by more than a factor of five (Fig. 9). The resultant increase in mass of sediments suspended in the estuary was accompanied by a significant alteration in composition. Inorganic materials dominated the suspended load despite a factor of three increase in the amount of particulate organic carbon in suspension (Fig. 11).

In addition to the variations in concentration and composition, major storm events serve to perturb the mass transport characteristics within the lower river. In this area, the sheltered, low energy conditions limit local resuspension of sediments. A major increase, therefore, is most probably the result of increased material suspension within adjacent Fishers Island Sound and subsequent shoreward transport into the Thames River. The progressive decrease in material concentrations with increasing distance from the Sound shown in Figure 9 is clearly representative of such a system. As a result of this phenomenon, significant volumes of new material are introduced into the estuary and regional deposition rates are increased. Given the magnitude of deposition inferred using average material concentrations, it is not unreasonable to conclude that a major fraction of the annual accumulation of sediments may be deposited following storm events.

Reviews of local meteorological data obtained over a ten year period indicate that storms similar in intensity to the February, 1976 event may occur as often as three times each year.

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Reviews of local meteorological data obtained over a ten year period indicate that storms similar in intensity to the February, 1976 event may occur as often as three times each year.

This rate is significantly higher than the normal frequency of required channel dredging. The Thames River was last dredged in 1965 and, typical of most northeastern U.S. estuaries, should require maintenance at a rate of approximately once every five to ten years. Although this rate is greater than the frequency of occurrence of major tropical storms (hurricanes), it is well below that of the more common coastal storm. When considered in combination with the relative magnitude of the response produced by dredging vis-a-vis storms, this fact suggests that the variability of the suspended material field in the lower Thames is primarily governed by aperiodic storm events. In comparison, dredging-related perturbations, at least on the short term, are negligible.

Summary

The suspended material field in the Thames River Estuary is characteristic of a sediment poor system and displays low average concentration levels (<5 mg/l) and erratic variability with no obvious spatial or temporal trends. In this system, the Phase 1 dredging operation:

1. Produced perturbations in suspended material concentrations and composition that were confined to an area within 300 to 500 yards of the operating dredge and barge.
2. Produced an increase in total suspended load within the estuary that was small in comparison to that produced by typical aperiodic storm events.
3. Caused no major alterations in mass transport within the estuary.

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B. THAMES RIVER HYDROGRAPHY, PHYTOPLANKTON, AND TRACE METAL CONCENTRATIONS
IN WATER, SEDIMENT AND SHELLFISH

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INTRODUCTION

This section summarizes trace metal analyses of three species of shellfish: Crassostrea virginica, Mercenaria mercenaria and Pitar morrhuana; surficial sediment; and water samples collected from Thames River, Connecticut during the first phase of the U.S. Navy's dredging project (July 1974-June 1976). Additional data on temperature, salinity, oxygen, chlorophyll a and c, gross pathological examination of the shellfish, as well as observations on mercury and chlorophyll a concentrations in the immediate vicinity of the dredge-barge are also presented. The study was originally designed to obtain predredging base line data in sufficient detail to permit discovery or evaluation of effects of dredging, if any, on the environment. Predredging data were obtained only for the month of July 1974, which rendered interpretations of certain data sets difficult.

HYDROGRAPHY

The temperature and salinity regime of Thames River typify a partially mixed estuary (Pritchard, 1955) which is characterized by more saline water at the bottom and less saline water at the surface.

Sampling was scheduled at 1-, 2- or 3-month intervals which

were dictated largely by weather conditions, particularly during the winter. Eight cruises were made to the River. During each cruise, temperature, salinity and oxygen measurements were taken routinely at the surface, mid-depth and one foot off bottom using a Hydrolab Temperature-Salinity-Oxygen probe at stations located on the six lateral transects (Fig. 1). In addition 64 water samples were also obtained, which were kept in coolers. Upon returning to the laboratory, these samples were processed immediately for chlorophyll a, b and c determinations according to the method of Strickland and Parsons (1968).

Temperature. Average water temperature from July 1974 to May 1976 varied from 20 to 24°C in the River (Table 1). The highest temperature was recorded in July while the lowest was observed in February. Furthermore, temperature stratification in the water column was most prominent during the month of July. In the colder months of the year, surface and bottom temperature approached uniform distribution with the bottom temperatures being slightly lower than those of the surface.

Salinity. Annual variations of salinity in the river are given in Table 1. Seasonal fluctuations were greater in surface waters than bottom waters. In July and October the maximum salinity of the surface water was 17 and 20 ppt respectively, while that of the bottom water was 25 and 30 ppt. Minimum salinities of 6 to 12 ppt were observed in February, April and May. These variations are probably associated with either spring runoff or heavy precipitation.

Figure 1. Water, sediment and shellfish sampling stations in Thames River, Connecticut.

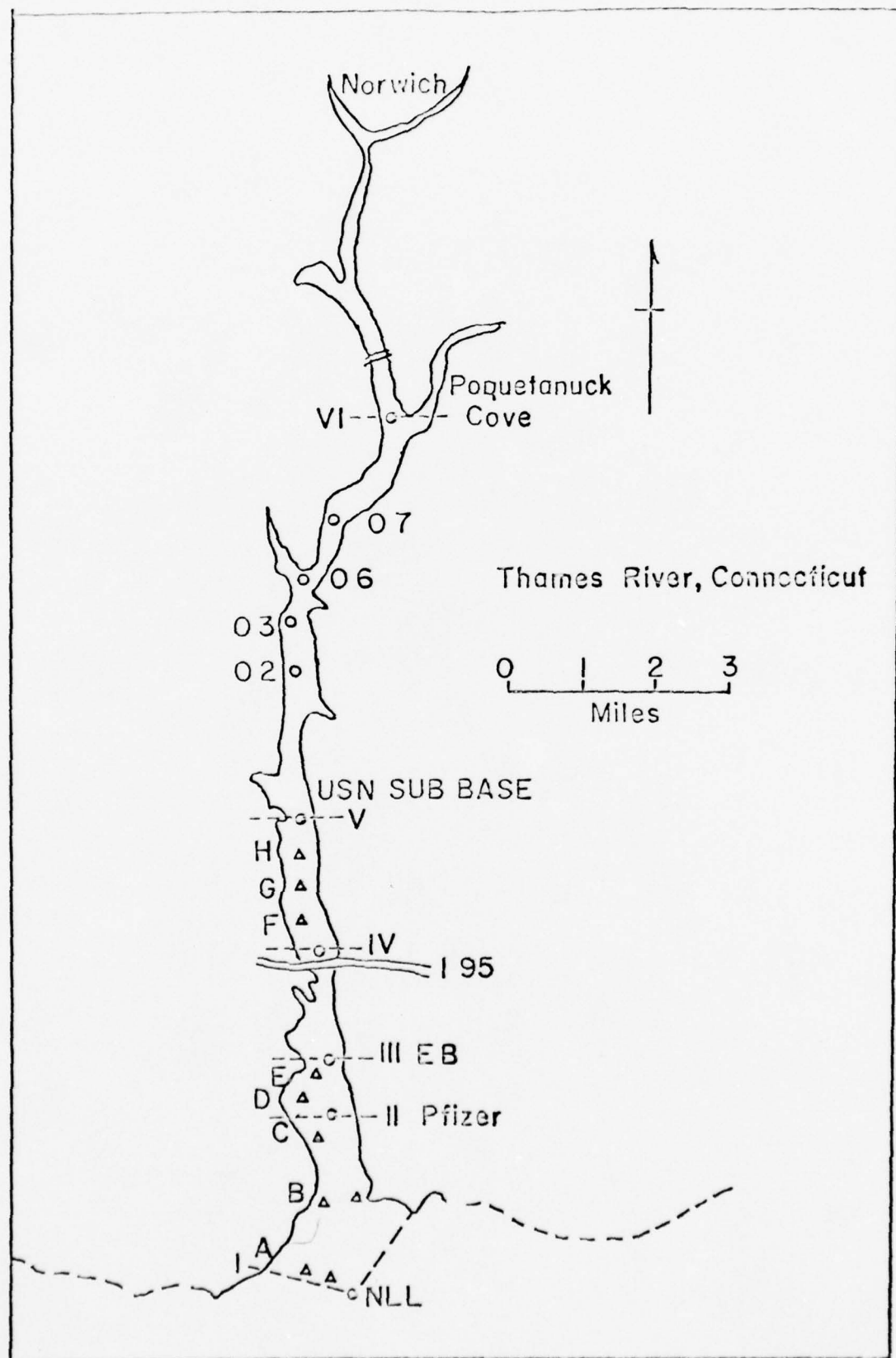


TABLE 1. Annual average temperature, salinity, oxygen, chlorophyll a and c variations observed in Thames River surface and bottom water.

		7/74	11/74	2/75	4/75	7/75	10/75	2/76	5/76
Temp.	S	20.6	14.3	2.4	7.8	24.1	15.8	4.6	13.6
	B	17.1	14.0	3.8	6.0	20.0	16.4	4.0	9.4
S ⁰ /oo	S	23.1	13.2	10.2	11.5	16.8	20.0	10.0	6.2
	B	26.4	22.8	24.2	20.5	24.8	29.5	21.8	24.7
O ₂	S	8.6	6.4	11.6	9.9	8.4	6.4	9.9	8.0
	B	5.9	6.5	10.3	11.0	5.6	6.4	10.2	8.2
Chlorophyll a	S	37.9	2.2	1.5	2.5	18.2	3.4	2.2	5.0
	B	12.6	1.8	2.0	2.3	8.5	2.6	3.2	4.6
Chlorophyll c	S	15.2	2.8	3.1	1.5	10.0	1.9	5.6	5.6
	B	6.2	1.8	3.6	1.2	7.8	2.0	3.6	10.4

Temp. in °C; S ‰ in ppt; O₂ in ppm; Chlorophyll a and c in mg/M³.

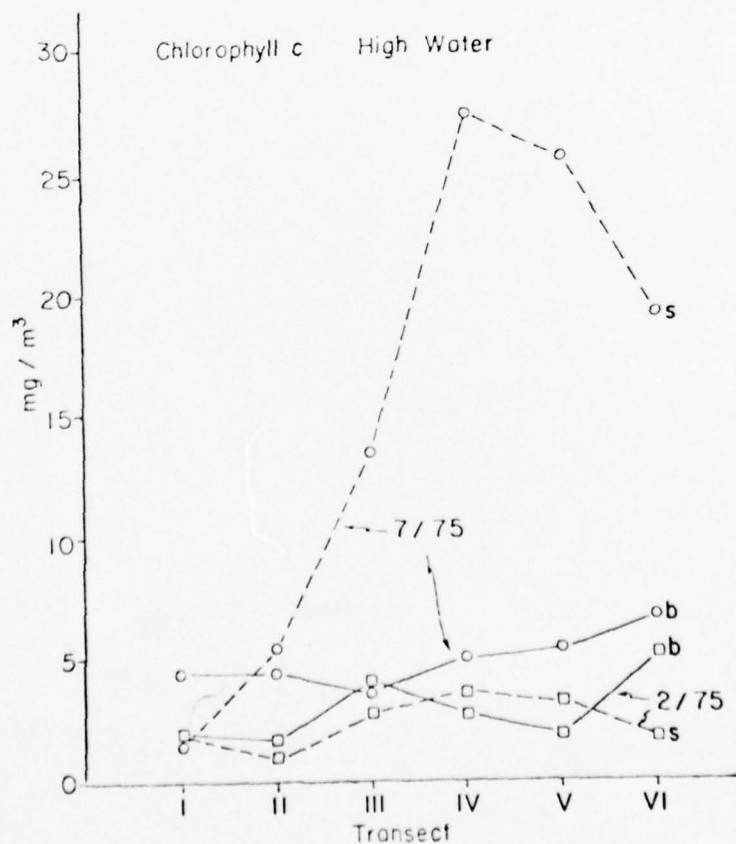
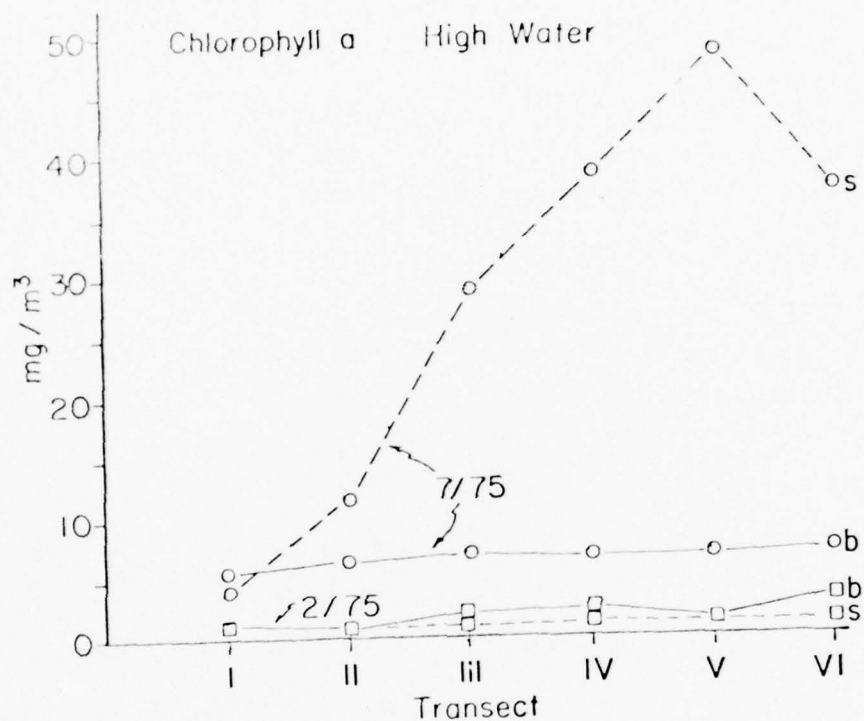
Oxygen. Oxygen concentrations of the surface and bottom water exhibited least variation in October, November, February, April and May. In July, oxygen content of bottom water was significantly lower than that of surface water; the stratification was greatest in Transect VI where oxygen content of less than 1 ppm was not uncommon in bottom water.

PHYTOPLANKTON

Chlorophyll a and c. The seasonal variations of chlorophyll a and c are presented in Table 1. There was no consistent difference between the chlorophyll measurements of the surface and bottom water for October, November, February, April and May; in general, chlorophyll a and c varies within narrow limits (1 to 5 mg/M³). Chlorophyll a (18-37 mg/M³) and c (10-15 mg/M³) were most abundant in July in surface water samples. Figure 2 shows that chlorophyll in the surface water sample decreased dramatically toward lower transects (I, II and III). It was noted that the concentration of chlorophyll a of July 1974 (predredging) was twice that of July 1975 (post dredging).

Determination of the effect of sediment elutriate on photosynthesis. In an attempt to determine whether elutriate from resuspension of dredge spoil could affect phytoplankton production, a sediment elutriate was made by pooling 10 gm of sediment from each of the three grab samples collected from transects I-V which are located from the mouth of the river to the U.S. Navy Submarine Base (Fig. 1). The sediments (150 gm) were mixed with two liters of sea water collected from the end of the Marine Research Laboratory pier. The sediment

Figure 2. The distribution of chlorophyll a and c in Thames River water on February and July 1975 during high water. "s" denotes surface water and "b" indicates bottom water.



suspension was shaken for 48 hrs. on a mechanical shaker and allowed to settle for six days. The elutriate was decanted, prefiltered and finally Millipore filtered (0.45 μ m).

Inhibition of photosynthesis was determined by introducing aliquots of the elutriate into five light and five dark bottles. Zooplankters were screened from the sea water. In the first series, one ml of elutriate was pipetted into each BOD bottle. Ten ml of elutriate were placed into each bottle in the second series. The two series were incubated ca. 30 cm below the surface off the Marine Research Laboratory dock with their appropriate controls which consisted of two pairs of light and dark bottles without elutriate. Five such runs were carried out in July and August 1974 to allow for different light conditions. The light intensity was determined at three different times during each run with a photometer at the surface and the percentage difference at 30 cm with that of the surface was determined with a submersible photometer.

After in situ incubation of the light and dark bottles, dissolved oxygen was determined by the Winkler method. Gross photosynthesis expressed as mg of carbon fixed per cubic meter per hour was determined by the method outlined in Strickland and Parsons (1968). The results from the five separate runs were analyzed statistically for differences by the use of a completely randomized block design. The results of the five runs are summarized in Table 2. The assumption that all samples within each run were homogeneous is valid, since during each run all sample bottles contained

TABLE 2 . Gross photosynthesis expressed as mg carbon fixed per cubic meter per hr in the control and experimental (sediment elutriate added) BOD bottles containing natural populations of phytoplankton.

Date	Light Intensity at 30 cm (lux)	% Light of Surface	Incubation Time (hr)	ml elutriate added	C mg/M ³ /hr
7/26	27,500	55	4	0	149.99
				1	138.65
				10	187.80
7/30	24,035	52	5	0	113.94
				1	110.92
				10	109.91
8/5	20,625	58	5	0	31.26
				1	41.34
				10	25.21
8/12	44,352	84	6	0	42.85
				1	26.89
				10	25.21
8/16	35,200	80	6.5	0	32.58
				1	26.37
				10	44.21

ANOVA

Sources of Error	d.f.	Sum of Squares	Mean Squares	F
Treatments	2	232.78	116.93	0.545 ns
Blocks	4	41047.65	10261.91	
Error	8	1708.92	213.61	
Total	14	42989.35		

$$F(0.05) = 4.46$$

water from the same source, incubated under the same light condition and analyzed at the same time. The results on the analysis of variance show that $F=0.545$, with 2,8 degrees of freedom, is not significant and that the null hypothesis is sustained (Table 2). It is concluded that the addition of sediment elutriate to a natural population of phytoplankton during July and August does not significantly influence photosynthesis.

TRACE METALS IN WATER SAMPLES

Water samples for trace metal analyses were obtained from each transect by separately pooling surface and bottom water. Thus, two water samples (one pooled surface and one pooled bottom water) were obtained from each transect during each phase of the tide. There were 8 cruises made and a total of 204 water samples was obtained for Zn, Cu, Cd, Ni, Pb and Hg analysis. These trace metals were determined by the APDC-MIBK extraction method of Brewer et al. (1969) with modifications developed in my laboratory.

The modifications include the following: The pH of 400 ml of previously acidified and Millipore filtered seawater ($0.45 \mu\text{m}$) was adjusted to 4.5 with a 5% Trizma base (Sigma). The Millipore filter used was soaked in 1% APDC solution overnight to eliminate the metal contaminants in the filter. This precaution is essential to eliminate erroneous results particularly in the cases of Pb and Zn which are present as universal contaminants. The nickel and lead contents of the extractions were determined by the use of a IL Model 455 Flameless Atomizer, and copper, zinc and cadmium by the standard

flame atomic absorption spectrophotometry using a IL Model 151 Atomic Absorption Spectrophotometer.

The linear regression equations for the absorbance vs. the concentrations of the unspiked and spiked samples of the first and second extraction are obtained by the least square method. The concentration of the metal in the sample can then be determined by solving the two simultaneous equations. A program was written to do the repetitive calculations on a Monroe Model 1655 programmable calculator.

The mercury concentrations of the water samples were determined by a cold trap preconcentration method developed by Fitzgerald et al. (1974). The water samples were acidified in the field and stored in acid cleaned Teflon bottles.

Table 3 summarizes the mean concentrations of nickel, lead, zinc, cadmium, copper and mercury in surface and bottom water samples collected from Thames River from July 1974 to May 1976. Zinc concentrations were the highest (11 to 28 ppb) among the six trace metals examined. Nickel concentrations were the second (3 to 10 ppb). Copper and lead concentrations ranked the third and fourth respectively and varied between 1 to 5 ppb, while most of the mean cadmium concentrations remained less than 1 ppb. The concentration of mercury was one order of magnitude lower than that of cadmium; the mean concentrations varied from 4 to 50 parts per trillion. Zinc and cadmium concentrations in Thames River are three times and nickel 5-10 times higher than the reported concentrations for Connecticut River (Fitzgerald et al. 1974b); copper and mercury concentrations are comparable for the two rivers

TABLE 3. Mean trace element concentrations found in the surface and bottom water samples collected from Thames River, Connecticut, July 1974 to May 1976. Hg is expressed in terms of parts per trillion (ppt) and all others ppb.

Trace Element	Layer		7/74	11/74	2/75	4/75	7/75	10/75	2/76	5/76
Ni	S	N	18	12	12	12	12	12	12	12
		Mean	4.35	5.08	6.07	9.92	3.65	6.84	6.34	5.57
		S.D.	3.02	0.93	4.55	18.18	2.05	5.78	3.45	3.40
	B	Mean	5.75	6.62	5.70	7.08	6.33	5.97	8.70	5.86
		S.D.	7.93	1.69	3.28	3.49	2.46	2.66	7.17	2.99
Pb	S	Mean	1.78	2.70	1.63	4.18	1.37	2.19	2.34	1.44
		S.D.	1.23	4.02	1.30	7.13	0.77	2.00	1.67	1.09
	B	Mean	1.76	5.36	1.16	1.42	1.44	1.67	3.37	1.10
		S.D.	1.40	6.99	1.19	1.38	0.97	2.19	5.28	1.08
Zn	S	Mean	27.62	21.97	20.40	11.20	13.52	26.63	14.90	11.74
		S.D.	17.71	20.02	18.88	3.59	5.90	19.85	9.84	6.65
	B	Mean	21.74	16.79	17.21	11.80	28.39	28.15	18.64	14.43
		S.D.	13.01	9.87	7.70	7.21	53.29	46.41	8.17	8.27
Cd	S	Mean	0.94	1.38	0.75	1.03	0.83	0.87	0.69	1.38
		S.D.	1.18	2.64	0.52	1.55	0.49	0.73	0.45	1.66
	B	Mean	0.49	1.99	1.11	0.46	0.99	0.74	0.98	0.80
		S.D.	0.74	4.60	0.96	0.32	1.33	0.64	0.91	0.84
Cu	S	Mean	2.22	1.17	4.11	2.66	4.38	4.21	5.25	3.66
		S.D.	1.12	1.17	6.85	1.65	2.93	2.93	3.28	2.69
	B	Mean	2.79	1.73	2.24	2.92	3.91	3.17	5.58	2.49
		S.D.	2.11	1.28	1.05	1.25	2.28	1.17	1.69	1.71
Hg	S	Mean	29.33	15.42	4.75	8.42	19.94	5.31	6.33	12.40
		S.D.	56.99	6.93	1.54	2.53	10.87	3.14	2.84	5.70
	B	Mean	51.56	16.33	5.42	4.75	27.47	7.72	10.22	7.66
		S.D.	95.03	10.92	2.31	2.73	17.87	5.38	3.83	4.54

(Fitzgerald et al. 1974b; Fitzgerald and Lyons, 1973). No lead data are available for Connecticut River.

Three-way analysis of variance (Time X Transect X Tide; Time X Transect X Layer) reveals significant variations in Zn, Ni, Pb, Cu, Hg and Cd with respect to time and transects (Table 4 and 5). It is also significant to note that in Ni, Pb, Cd and Hg the highest concentrations were obtained either before or during dredging (Table 3). Copper concentrations were significantly higher in the post dredging period (July 1975 to May 1976) than in the pre-and during dredging period. Concentrations of Ni, Cu and Pb were significantly influenced by the tide. Copper was the only trace metal to show significant variations in surface and bottom water during low tide. Variations of Ni, Pb, Cu and Hg concentrations with respect to the transect are shown in Table 6. In general, Ni, Pb and Cu concentrations were higher in the upper river water samples (Transect IV, V and VI) than in the lower river samples suggesting the possibility that sources of these trace metals might be located in the upper river. Mercury levels in the surface water, on the other hand, were significantly higher in the lower river (Transect I, II and IV) than in the upper river transects indicating a possible lower river origin of this trace metal. In view of the evidence provided by our field studies on the distribution and concentration of Hg in the immediate vicinity of the dredge-barge (see the section below), we could conclude that the contribution of Hg in the lower river attributable to the dredging operation, if any, was probably very small.

TABLE 4. Summary of significant results on 3-way analysis of variance (Time X Transect X Tide) for water samples collected from Thames River, Connecticut, July 1974 to May 1976. (Surface water, N=102; Bottom Water, N=102).

Trace Element	Source	d.f.	F	Depth
Ni	Time	7,6	6.055**	Surface
	Transect	5,6	8.564*	
	Tide	1,6	11.389*	
Pb	Time	7,6	6.170*	Surface
	Transect	5,6	8.573*	
	Tide	1,6	1.322	
Cu	Time	7,6	20.581***	Surface
	Transect	5,6	16.857***	
	Tide	1,6	11.333*	
Hg	Time	7,6	13.089***	Surface
	Transect	5,6	6.089*	
	Tide	1,6	5.634	
Pb	Time	7,6	15.500***	Bottom
	Transect	5,6	8.808**	
	Tide	1,6	9.577*	
Cd	Time	7,6	4.318*	Bottom
	Transect	5,6	6.470*	
	Tide	1,6	1.303	

*, P < 0.05; **, P < 0.01; ***, P < 0.001.

TABLE 5. Summary of significant results on 3-way analysis of variance (Time X Transect X Layer) for water samples collected from Thames River, Connecticut, July 1974 to May 1976. (High water, N=108; Low Water N=96).

Trace Element	Source	d.f.	F	Tide
Pb	Time	8,40	4.20***	High Water
	Transect	5,40	7.15***	
	Layer	1,40	0.43	
Zn	Time	8,40	6.23***	High Water
	Transect	5,40	2.03	
	Layer	1,40	0.002	
Cu	Time	8,40	2.50*	High Water
	Transect	5,40	2.80*	
	Layer	1,40	0.003	
Hg	Time	8,40	4.91***	High Water
	Transect	5,40	0.7	
	Layer	1,40	2.29	
Pb	Time	7,35	2.25	Low Water
	Transect	5,35	2.84*	
	Layer	1,35	0.84	
Cu	Time	7,35	15.24***	Low Water
	Transect	5,35	3.72**	
	Layer	1,35	4.88*	
Hg	Time	7,35	19.78***	Low Water
	Transect	5,35	12.02***	
	Layer	1,35	2.69	

*, P < 0.05; **, P < 0.01; ***, P < 0.001; Layer denotes surface and bottom water.

TABLE 6. Mean trace element concentrations in Thames River water samples, which show significant variations among various transects.

		Surface Water				Bottom Water	
Transect		Ni *	Pb *	Cu ***	Hg **	Pb ***	Cu *
I	Mean	6.83	1.44	4.65	13.98	1.49	0.60
	S.D.	4.95	1.62	5.62	9.68	2.22	0.59
II	Mean	4.26	1.69	2.44	24.99	1.75	0.67
	S.D.	3.00	0.90	2.22	59.39	2.95	0.68
III	Mean	5.14	1.80	1.91	9.68	1.38	0.87
	S.D.	2.53	1.48	1.65	7.11	1.74	0.83
IV	Mean	3.91	1.51	3.52	13.26	1.87	1.05
	S.D.	2.31	0.90	2.24	11.28	1.43	0.90
V	Mean	6.41	3.53	3.68	7.68	1.92	1.80
	S.D.	4.04	5.89	2.61	3.97	1.16	3.95
VI	Mean	8.75	3.10	4.10	11.27	4.41	0.52
	S.D.	15.32	3.72	3.45	9.74	6.82	0.75

*, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$.

Multiple regression analysis of temperature, salinity, oxygen, chlorophyll a, b and c and the six trace metals in water samples revealed a number of significant correlation coefficients (Table 7). Mercury concentrations are significantly correlated with both temperature and chlorophyll. These relationships can be expressed in the following two regression equations:

$$Y = 1.198 X + 1.080,$$

where Y = mercury concentration and X = temperature (F=10.252; d.f. 1,195; P <.01), and

$$Y = 0.993 X + 10.936,$$

where Y = mercury concentration, X = chlorophyll c (F=6.108, d.f. 1,195; P <0.05). The seasonality of mercury concentration suggested by the first regression equation is further corroborated by data presented in Table 3; highest mercury concentrations were found in July and lowest levels in colder months of the year. The second regression equation strongly suggests that the high mercury concentration is associated with phytoplankton which is also temperature dependent (Table 1 and 7).

Copper concentrations are negatively correlated with salinities; this relationship is represented by the following regression equation:

$$Y = -0.048 X + 4.154,$$

where Y = Cu concentrations and X = salinity (F=4.978; d.f. 1,195; P <0.05). Similar observation was reported by Fitzgerald et al. (1974b) in the Connecticut River water.

TABLE 7. Multiple regression analysis of temperature, salinity, oxygen, chlorophyll a, b and c and trace metal concentrations observed in water samples collected from Thames River, Connecticut, July 1974 to May 1976 (N=204).

	S ⁰ /oo	O ₂	A	B	C	Ni	Pb	Zn	Cd	Cu	Hg
Temp	.2583**	-.1499	.4943***	.0621	.3675***	-.1519*	-.0391	.1341	.0294	-.0803	.2188**
S ⁰ /oo		.2283**	.0187	-.0951	.0135	.1151	-.1074	.1084	-.0582	-.1772*	.1414
O ₂			.0418	-.0849	.0735	.0376	.0031	-.0123	-.0204	-.0046	-.0739
A				.3589***	.6779***	-.0668	-.0274	.1220	-.0049	-.0040	.1566*
B					.1863**	.0538	-.0076	-.0095	-.0198	.1072	-.0526
C						.0514	-.0146	.0598	-.0094	-.0126	.1870**
Ni							.1286	.0961	.0208	.2001**	-.0669
Pb								.0593	-.0055	.0404	-.0245
Zn									.0004	.0014	-.0071
Cd										.0627	-.0078
Cu											-.0596

B-17

*, P < 0.05; **, P < 0.01; ***, P < 0.001.

DISTRIBUTIONS OF CHLOROPHYLL A AND MERCURY CONCENTRATIONS
IN THE IMMEDIATE VICINITY OF THE DREDGE-BARGE

Two high resolution surveys in the immediate vicinity of the dredge-barge during dredging operations were conducted in the river, on April 9 and May 14, 1975. These studies were conducted concurrently with Dr. Bohlen's suspended load investigations when the dredge-barge was located at the mouth of the Thames and the State Pier respectively. During the two cruises, a total of 40 stations was occupied and 126 water samples were collected for chlorophyll a and mercury determinations.

Chlorophyll a and mercury concentrations of samples obtained on April 9 are shown in Figures 3 and 4; the dredge-barge was located at the southernmost boundary of the channel. It is significant to note that the levels of chlorophyll a and mercury are significantly lower in the plume (south of the dredge-barge) than in the background (north of the dredge-barge).

The investigation carried out on May 14 was conducted approximately 500 yd. south of the drawbridge in the vicinity of the State Pier which was the northernmost boundary of the Phase I project. The data are presented graphically in Figures 5 and 6. Concentrations of chlorophyll a were slightly higher in the immediate vicinity of the dredge-barge at all three depths and quickly dissipated into the background levels within 250 yd from the site of dredging.

Although pockets of high mercury levels were observed, the general pattern of lower mercury concentrations in the immediate vicinity of the barge was detected again in this study. It was most prominent

Figure 3. The distribution of chlorophyll *a* concentrations (mg/M³) in the vicinity of the dredge-barge (B) on April 9, 1975 in New London Ledge (NLL) area.

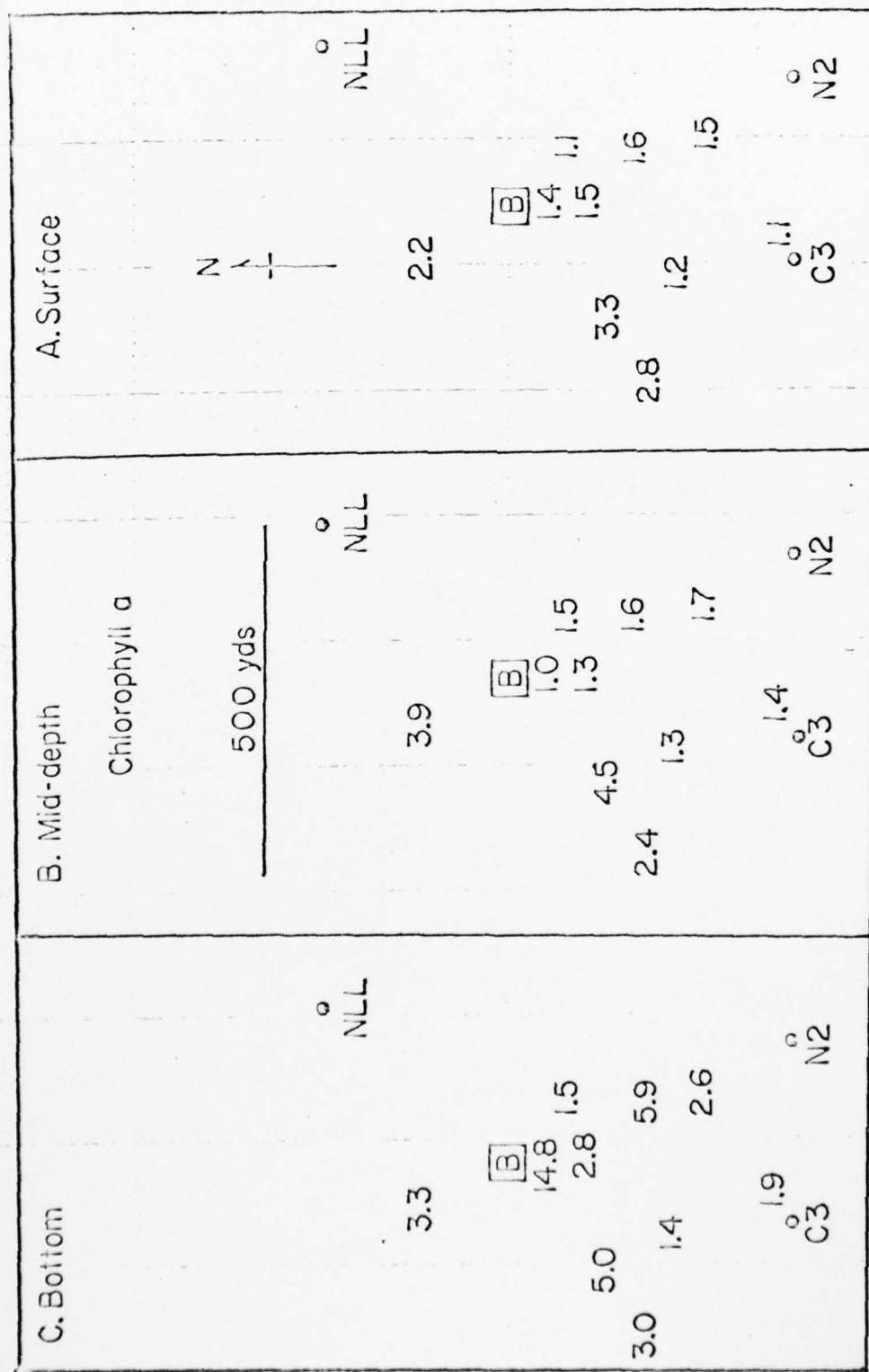


Figure 4. The distribution of mercury concentrations (parts per trillion) in the vicinity of the dredge-barge (B) on April 9, 1975 in New London Ledge (NLL) area.

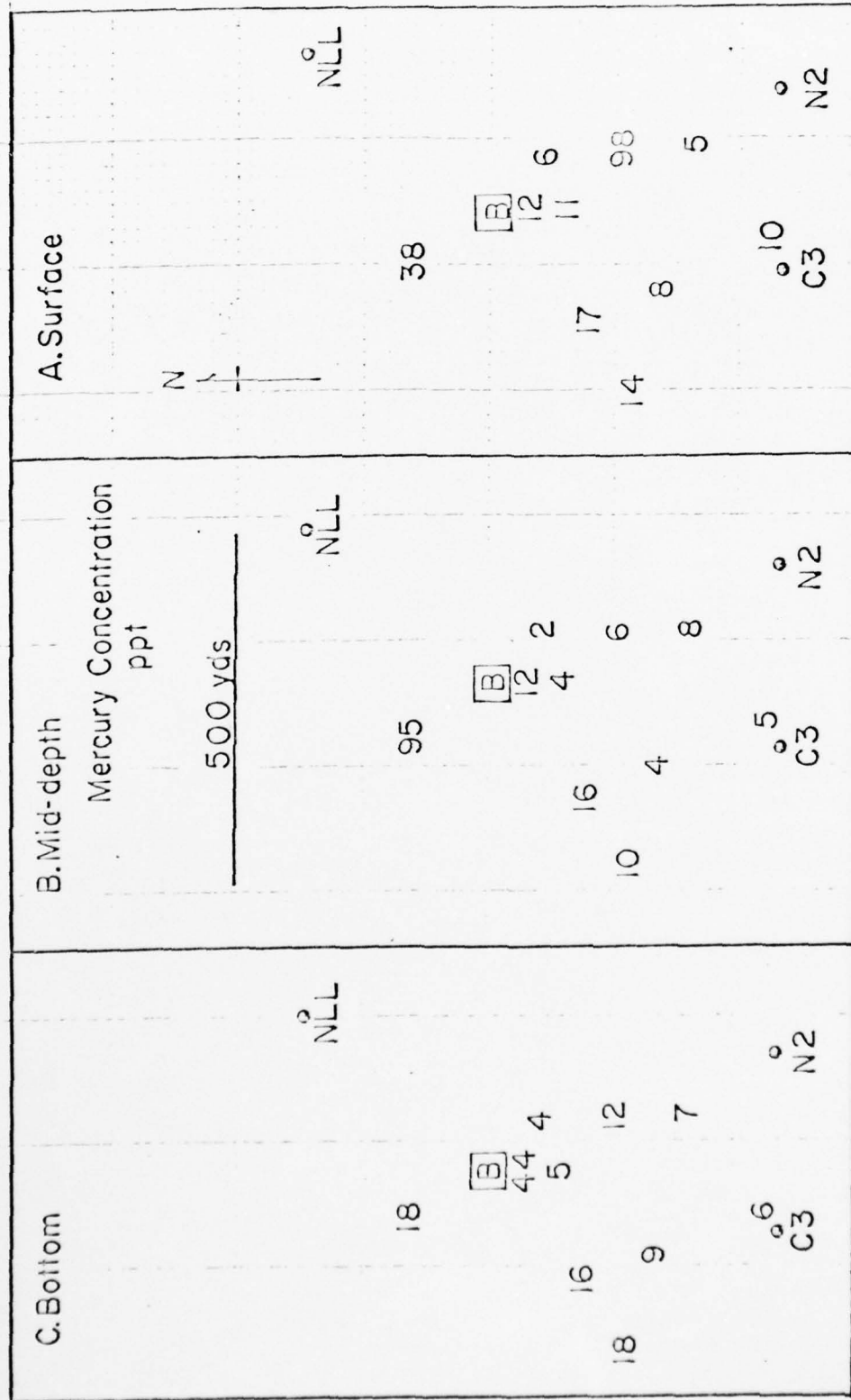


Figure 5. The distribution of chlorophyll *a* concentrations (mg/M³) in the vicinity of the dredge-barge (B) on May 14, 1975 in the State Pier (S.P.) area.

A. Surface



B. Mid-depth

Chlorophyll *a*

500 yds

C. Bottom

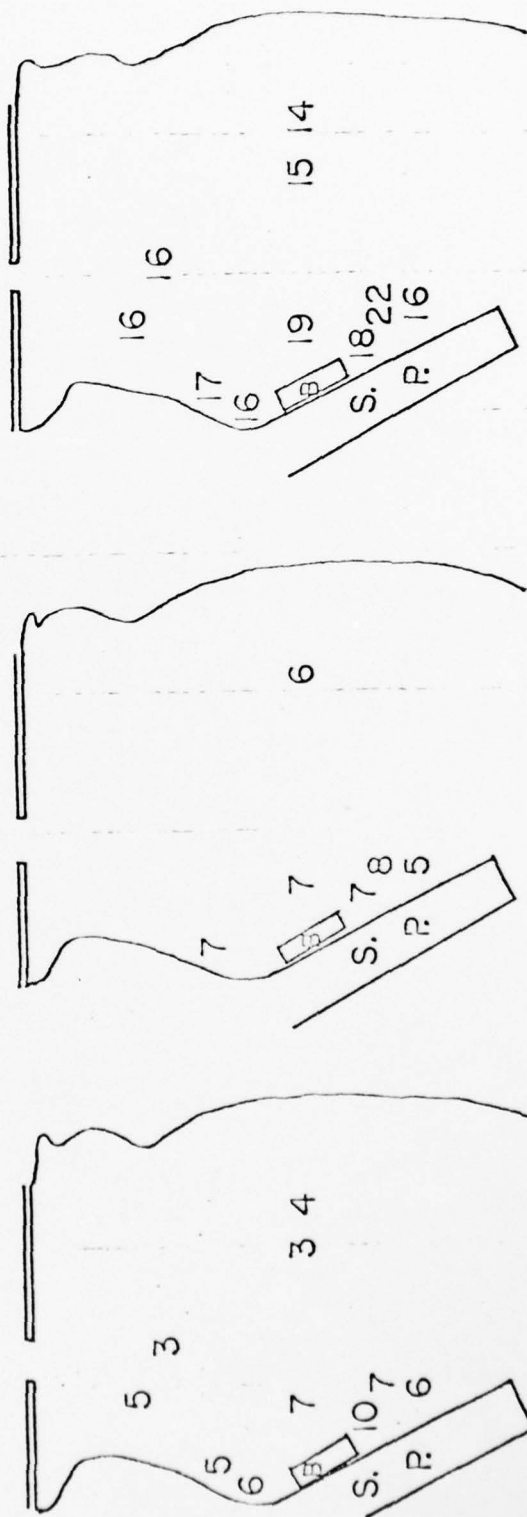


Figure 6. The distribution of mercury concentrations (parts per trillion) in the vicinity of the dredge-barge (B) on May 14, 1975 in the State Pier (S.P.) area.

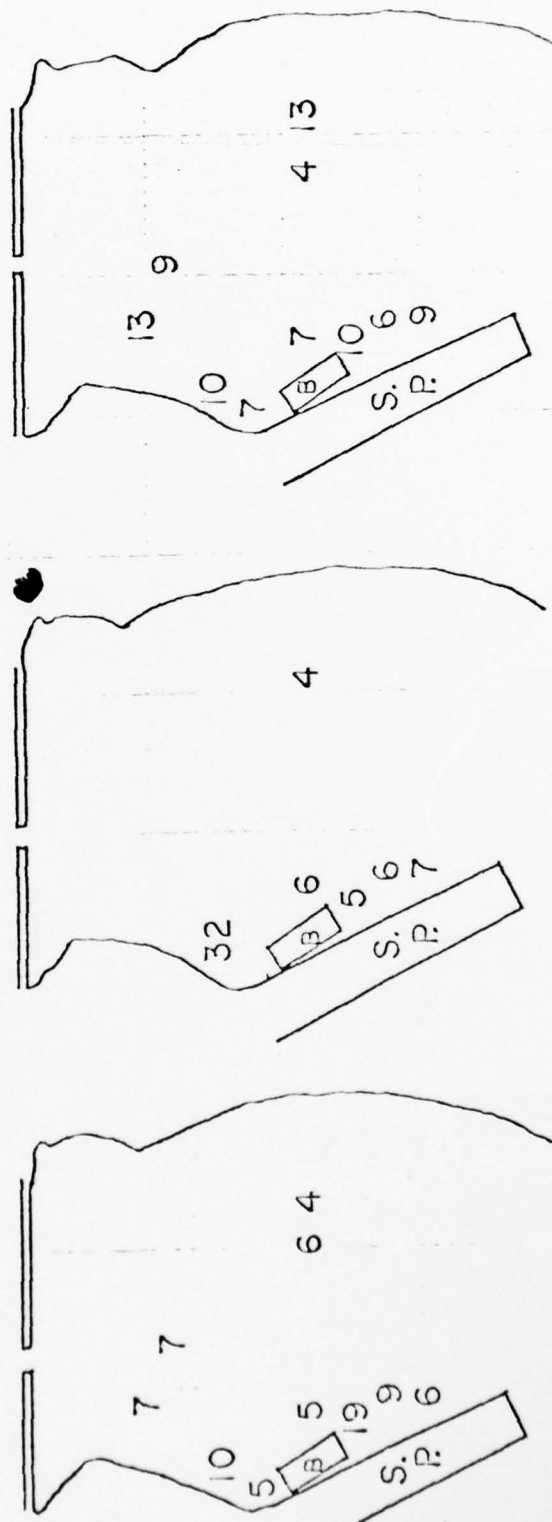
A. Surface

B. Mid-depth

C. Bottom

Mercury Concentration ppt

500 yds



in the surface and bottom samples.

Fields of lower chlorophyll a figures were observed at all three depths south of the barge on April 9, when the background concentration of chlorophyll a was approximately 2.2 mg/M^3 ; this observation suggested a possible transient interruption of photosynthesis in the water column presumably due to shading caused by a temporary increase in suspended load. During the cruise of May 14, however, the effect of dredging on photosynthesis appeared to be negligible, when the concentration of chlorophyll a in the surface samples on this date reached 14 to 16 mg/M^3 . Apparently the productivity in the river was such that the small effect of increased suspended load on photosynthesis was submerged in the high background level of chlorophyll a. There were indications that chlorophyll a concentrations were slightly higher in the immediate vicinity of the barge.

The apparent decrease of mercury concentrations in the vicinity of the barge was perhaps associated with the adsorption of the trace metal on the resuspended sediment which quickly settled out of the water column, although experimental evidence is still lacking. Similar reduction of mercury values during dredging has been documented by Jeane and Pine (1975). A second possible explanation for the lower mercury concentrations observed is the release of fulvic acids from the resuspension of sediment; fulvic acids are known to be present in the sediment and furthermore, they bind trace metals readily (Newman and Feng, unpublished data). Since the technique employed for mercury determination detects only reactive forms or readily available mercury in the water sample, the organically bound mercury may have escaped detection.

TRACE METALS AND ORGANIC CARBON IN SURFICIAL
SEDIMENT SAMPLES

Three grab samples from each transect in the ship's channel were taken on July 18, 1974 and February 20, 1976; 36 samples were obtained. The 1974 samples provide a predredging background for the geochemical condition of the channel sediment, while the 1976 samples portray the post-dredging trace metal profile.

For the analysis of Ni, Cd, Cu, Zn and Pb in sediment samples the following procedures were carried out at the laboratory. Twenty grams of wet samples were removed from the top 10 mm of the sediment samples, dried to a constant weight in an oven at 60° C. One half grams of a pulverized dry sediment were extracted with 10 ml concentrated nitric acid (15 N) in a 25 ml screw capped volumetric flask for 6 hours. The extract was filtered through a precleaned glass fiber filter (Gilman) and diluted to 25 ml with distilled-deionized water. The solutions were then analyzed by the established procedures of atomic absorption spectrophotometry. For mercury analysis, 0.5 gm of wet sediment were digested with 30 ml of sulfuric acid (36 N) and processed as recommended for samples of shellfish tissues according to the procedure outlined in the Perkin-Elmer Instruction Manual for Mercury Analysis System (303-3119). The results were expressed in terms of micrograms per gm dry weight of sediment.

Determinations of percent organic carbon in the sediment samples were carried out by the method of Gaudette et al. (1974). The mean percent organic carbon content as well as the mean concentration of

Ni, Cd, Cu, Zn, Pb and Hg derived from triplicate samples in the sediment from the six transects on two sampling dates are presented in Table 8 and 9. The concentrations of the six trace metals and organic carbon in the channel sediment tend to increase toward upriver. As would be expected, areas of high concentrations are generally associated with industrial activities along the river, e.g., Transect II, III and V correspond to Pfizer, Electric Boat and the U.S. Submarine Base respectively. With the exception of nickel, the highest concentration of the trace metals and organic carbon was found in the sediment of Transect VI; these results are highly significant as suggested by the two-way analysis of variance. The channel from which the Transect VI sediment samples were taken, is characterized by low oxygen content near the bottom, particularly during the summer. It is assumed that the concentration of trace metals at this transect is probably enhanced by the sewage from the City of Norwich upriver. When one compares the predredging data with that of the postdredging period, there is a discernible reduction of copper ($P < 0.05$) in the lower river transects in postdredging samples. The organic carbon content, on the other hand, shows a slight increase in the post-dredging data which is statistically significant (Table 8, $P < 0.01$). The data suggest the possible existence of a seasonal cycle in the organic carbon of sediments; it is postulated that the slightly higher organic carbon content in the coldest month (February) of the year is due to the reduced rate of organic carbon breakdown by microbial activities.

When the trace metals and percent organic carbon data are subjected to multiple regression analysis, it is revealed that the

TABLE 8. Mean percent organic carbon in surficial sediment samples obtained from the ship's channel in Thames River, Connecticut, on July 2, 1974 and February 20, 1976. Mean is derived from triplicate samples.

Transect	Mean % Organic C \pm 1S.D.	
	7/2/74	2/20/76
I	1.61 \pm 0.07	1.65 \pm 0.24
II	2.58 \pm 0.38	2.61 \pm 0.19
III	2.45 \pm 0.32	3.09 \pm 0.36
IV	2.96 \pm 0.19	2.89 \pm 0.21
V	3.83 \pm 0.53	4.66 \pm 0.65
VI	7.23 \pm 0.48	8.14 \pm 0.81

ANOVA

Source	d.f.	Sum of Squares	Mean Square	F	P
Dates	1	1.4161	1.4161	7.876**	<0.01
Transects	5	138.9120	27.7824	154.514***	<0.001
Dates X Transects	5	1.5051	0.3010	1.674	
Error	24	4.3153	0.1798	0.424	

** , P <0.01; *** , P <0.001

TABLE 9. Pb, Zn, Cd, Cu, Hg and Ni concentrations in surficial sediment samples obtained from the ship's channel in Thames River, Connecticut, on July 2, 1974 and February 20, 1976. The result is expressed as mean ppm \pm 1. S.D.

Transect	Pb		Zn	
	7/2/74	2/20/76	7/2/74	2/20/76
I	30.44 \pm 2.59	23.65 \pm 1.94	101.72 \pm 7.05	95.96 \pm 56.64
II	44.79 \pm 4.21	32.41 \pm 4.84	207.49 \pm 31.76	226.13 \pm 66.82
III	87.84 \pm 53.86	45.67 \pm 8.35	221.32 \pm 76.33	247.92 \pm 55.14
IV	46.46 \pm 14.12	34.39 \pm 23.91	161.91 \pm 55.00	218.21 \pm 171.23
V	69.46 \pm 35.45	71.47 \pm 3.73	255.04 \pm 108.49	359.89 \pm 15.85
VI	167.60 \pm 21.62	150.42 \pm 19.46	428.74 \pm 75.62	810.89 \pm 196.70
ANOVA	F		F	d.f.
Dates	3.984		9.579**	1,24
Transects	28.229***		22.087***	5,24
Transect	Cd		Cu	
	7/2/74	2/20/76	7/2/74	2/20/76
I	0.66 \pm 0.16	0.71 \pm 0.11	13.71 \pm 0.78	11.25 \pm 1.87
II	1.41 \pm 0.84	1.45 \pm 0.19	21.18 \pm 2.00	14.17 \pm 3.57
III	1.57 \pm 0.10	1.77 \pm 0.16	25.34 \pm 9.76	20.87 \pm 1.04
IV	1.26 \pm 0.29	1.48 \pm 0.24	20.31 \pm 6.94	16.66 \pm 12.02
V	2.29 \pm 0.52	2.17 \pm 0.05	34.78 \pm 20.52	36.13 \pm 3.03
VI	4.84 \pm 0.60	4.45 \pm 0.51	113.97 \pm 20.02	84.65 \pm 11.42
ANOVA	F		F	d.f.
Dates	0.0001		4.903*	1,24
Transects	74.363***		60.668***	5,24
Transect	Hg		Ni	
	7/2/74	2/20/76	7/2/74	2/20/76
I	.135 \pm .031	.111 \pm .041	17.09 \pm 1.48	17.00 \pm 0.80
II	.126 \pm .013	.089 \pm .004	43.65 \pm 6.82	38.12 \pm 4.42
III	.142 \pm .052	.184 \pm .108	57.11 \pm 19.80	47.79 \pm 13.98
IV	.198 \pm .058	.200 \pm .130	31.12 \pm 9.14	38.13 \pm 17.16
V	.237 \pm .118	.181 \pm .030	42.16 \pm 13.04	45.95 \pm 5.56
VI	.725 \pm .064	.729 \pm .130	45.70 \pm 6.71	51.45 \pm 4.47
ANOVA	F		F	d.f.
Dates	0.190		0.006	1,24
Transects	53.984***		8.867***	5,24

*, P < 0.05; **, P < 0.01; ***, P < 0.001.

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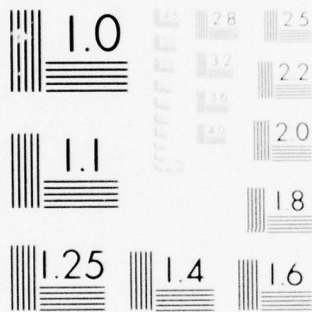
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metal concentration, with the exception of nickel, is correlated with the percent organic carbon content in the sediment (Table 10 and 11). The linear regression equations which describe these relationships, are summarized as follows:

1. Zn, $Y = 102 X - 68$, $r = 0.89$,
2. Pb, $Y = 19.94 X - 16.96$, $r = 0.96$,
3. Cu, $Y = 11.81 X - 14.58$, $r = 0.96$,
4. Cd, $Y = 0.54 X - 0.07$, $r = 0.98$, and
5. Hg, $Y = 0.09 X - 0.098$, $r = 0.84$,

where Y = [trace metal] and X = % organic carbon. This finding is significant in that by simply determining the percent organic carbon of the channel surficial sediment, it is possible to predict the five trace metals contained in the sample.

TRACE METAL CONCENTRATIONS IN THE THREE SPECIES OF SHELLFISH

The propensity of bivalve molluscs to accumulate waterborne contaminants, e.g., viruses, bacteria, toxic algae, trace metals, hydrocarbon, pesticides and related compounds from the environment, is well known. In a two-year study of the distribution and concentration of five trace metals (zinc, copper, cadmium, mercury and manganese) in oysters along the north shore of Long Island Sound, Feng and Ruddy (1975) report that the distribution and concentration pattern of zinc, copper and cadmium in the oyster coincide with the industrial and population centers of the coastal region of Connecticut. Recent work by Alexander and Young (1976) suggests that Mytilus californianus is a particularly sensitive indicator for copper, chromium and silver of urban point sources in southern California waters.

TABLE 10. Interelement and percent organic carbon correlations (r) in surficial sediment samples obtained from the ship's channel in Thames River, Connecticut, July 2, 1974.

	Pb	Zn	Cd	Cu	Hg	%C	Mean	S.D.
Ni	.3719	.6485	.3876	.3867	.2019	.3528	39.47	15.93
Pb		.7996	.8539	.8608	.8635	.8265	74.43	52.66
Zn			.8810	.9188	.8187	.8904	229.37	118.86
Cd				.9494	.9154	.9433	2.01	1.45
Cu					.9433	.9618	38.22	37.04
Hg						.9410	0.26	0.22
%C						---	3.44	1.90

Correlation coefficients (r) within the boundary are highly significant ($P < 0.01$; $N=18$).

TABLE 11. Interelement and percent organic carbon correlations (r) in surficial sediment samples obtained from the ship's channel in Thames River, Connecticut, February 20, 1976.

	Pb	Zn	Cd	Cu	Hg	%C	Mean	S.D.
Ni	.6054	.6458	.6344	.5846	.3450	.6132	39.74	11.16
Pb		.9462	.9427	.9959	.8631	.9629	59.66	45.94
Zn			.8845	.9488	.8438	.8928	326.50	255.36
Cd				.9431	.8707	.9757	2.01	1.23
Cu					.8643	.9654	30.62	26.85
Hg						.8449	0.25	0.24
%C						---	3.84	2.22

All correlation coefficients (r) except one (Ni-Hg), are highly significant ($P < 0.01$; $N=18$).

In the present investigation, attempts were made to monitor six trace metals in three indigenous species of bivalve molluscs, Crassostrea virginica, Mercenaria mercenaria and Pitar morrhuana, as a means of assessing the possible effect on these bivalves of trace metals released from the dredging operation.

A preliminary survey of the distribution of shellfish in the river showed that Pitar morrhuana and Mercenaria mercenaria occupied more or less the same range and were most abundant in the area 1.1 to 4.8 nautical miles north of the New London Ledge Light; 8 stations from A to H were designated for sampling M. mercenaria and P. morrhuana (Fig. 1). Major oyster (Crassostrea virginica) seed beds were located 6.8, 7.4, 8.2 and 9.0 nautical miles from the mouth of the river and designated as stations 0-2, 0-3, 0-6 and 0-7 respectively.

Sample collection. Mercenaria mercenaria and P. morrhuana were obtained by a hydraulic dredge, while C. virginica were collected by a standard oyster dredge employed by the local shellfishermen. Over 900 M. mercenaria and Pitar morrhuana each, and 264 C. virginica were sampled from July 1974 to May 1976.

Methods. Shellfish were cleaned of sediment and fouling organisms by brushing under running tap water and carefully denuded. In general the meats from 8 oysters or hard clams and 14 Pitar were pooled and homogenized without adding any diluent in a Tekmar homogenizer. The homogenized samples were lyophilized overnight using a Virtis freeze dryer. Zinc, copper, cadmium, lead and nickel in 0.4 gm of freeze-dried shellfish tissues were analyzed using the method of

Feng and Ruddy (1975). Only 0.2 gm of tissues were needed for mercury analysis which was accomplished by the method of Hatch and Ott (1968) using the principle of cold vapor atomic absorption spectrophotometry on a Coleman MAS-50 Mercury Analyzer.

Gross pathological survey of the shellfish was accomplished by dissecting 8 to 10 individuals of each species of shellfish from each station. The following anatomical regions were carefully inspected for gross pathological conditions: inner and outer aspects of gills, palps, mantle and pericardial cavity.

The average trace metal concentrations in the tissues of C. virginica, M. mercenaria and P. morrhuana obtained from various stations on a seasonal basis from July 1974 to May 1976 are summarized in Table 12, 13 and 14.

Temporal and spatial distribution of trace metals in shellfish.

Of the six trace metals detected, zinc was found to have the highest concentrations in the three species of bivalve molluscs (Table 12 and 13). In the oyster tissue the concentration of zinc was two orders of magnitude higher than that of M. mercenaria and P. morrhuana. The total mean concentration of copper in C. virginica was one order of magnitude higher than that of M. mercenaria and P. morrhuana, (Table 12). Copper concentrations in C. virginica exhibited a seasonal high in July, as did the zinc concentrations. In M. mercenaria and P. morrhuana, although the variations in copper concentration obtained on different dates were considered significant ($P < 0.001$), no

Table 12. Significant temporal variations in mean concentrations of zinc, copper, and nickel found in *Crassostrea virginica* (CV), *Mercenaria mercenaria* (MM) and *Pitar Morrhua* (PM) collected from Thames River, Connecticut, July 1974 to May 1976.

Trace Metal Species		7/74	11/74	3/75	5/75	7/75	12/75	3/76	5/76
Zn	CV	Mean	18978	14213	14482	16225	18884	8683	9114
		S.D.	1896	742	2978	3240	4070	450	2125
	MM	Mean	186	194	218	236	311	212	128
		S.D.	36	97	19	31	161	50	24
	PM	Mean	355	438	364	392	356	399	230
		S.D.	191	193	122	70	108	161	79
Cu	CV	Mean	1350	740	921	803	1023	579	277
		S.D.	127	24	204	181	222	357	66
	MM	Mean	30	24	23	25	24	17	23
		S.D.	5	4	4	2	5	5	7
	PM	Mean	20	17	14	20	16	10	16
		S.D.	2	6	2	6	3	2	4
Ni	CV	Mean	5.56	5.37	5.18	6.09	6.20	8.02	4.20
		S.D.	0.94	1.07	1.05	0.78	1.94	0.91	0.22
	MM	Mean	8.20	8.55	5.49	8.47	7.35	13.11	8.60
		S.D.	1.75	1.94	1.30	1.56	1.42	2.44	1.62
	PM	Mean	7.66	8.79	5.15	8.48	7.17	10.00	9.20
		S.D.	0.24	2.42	0.33	0.51	0.29	1.50	1.06

Table 13. Significant temporal variations in mean concentrations of cadmium, lead and mercury found in *Crassostrea virginica* (CV), *Mercenaria mercenaria* (MM) and *Pitar morrhuana* (PM) collected from Thames River, Connecticut, July 1974 to May 1976.

Trace Metal	Species	7/74	11/74	3/75	5/75	7/75	12/75	3/76	5/76
Cd	CV	Mean	6.84	3.11	5.48	5.90	8.12	2.78	4.18
		S.D.	1.53	0.52	0.78	0.62	1.70	0.44	0.57
	MM	Mean	1.08	0.67	1.18	1.69	2.02	1.11	2.01
		S.D.	0.19	0.35	0.23	0.30	0.16	0.50	0.21
	PM	Mean	*	*	*	*	*	*	*
		S.D.	*	*	*	*	*	*	*
Pb	CV	Mean	1.70	1.70	1.70	4.16	1.70	3.01	1.70
		S.D.	0.00	0.00	0.00	2.04	0.00	0.92	0.00
	MM	Mean	1.70	1.70	1.70	8.01	1.70	5.62	3.35
		S.D.	0.00	0.00	0.00	1.94	0.00	1.34	1.24
	PM	Mean	18.09	33.78	35.67	44.46	36.39	29.62	23.10
		S.D.	5.66	11.12	8.23	5.85	15.00	7.59	6.01
Hg	CV	Mean	0.39	0.21	0.36	0.28	0.28	0.26	0.24
		S.D.	0.02	0.07	0.02	0.04	0.06	0.03	0.02
	MM	Mean	0.35	0.32	0.23	0.21	0.22	0.26	0.03
		S.D.	0.10	0.14	0.07	0.07	0.07	0.11	0.07
	PM	Mean	0.19	0.22	0.22	0.12	0.16	0.18	0.20
		S.D.	0.05	0.03	0.02	0.02	0.01	0.04	0.01

Table 14. Significant spatial variations in mean concentrations of zinc, copper, nickel, lead and mercury found in *Crassostrea virginica* (CV), *Mercenaria mercenaria* (MM) and *Pitar morrhuana* (PM) collected from Thames River, Connecticut, July 1974 to May 1976.

Species	Station	Zn	Cu	Ni	Cd	Pb	Hg
MM	A+B+C	170±046	20.04±6.32	10.14±2.23	1.58±0.40	2.62±1.39	0.16±0.05
	D	239±091	18.67±4.24	9.16±2.09	1.53±0.56	2.90±2.06	0.20±0.04
	E	195±035	21.36±3.90	9.10±2.23	1.46±0.65	3.20±2.32	0.27±0.09
	F	264±144	24.98±4.82	7.58±1.72	1.44±0.62	3.55±2.47	0.25±0.06
	G	242±067	25.79±2.15	7.37±2.11	1.30±0.48	3.90±3.08	0.29±0.04
	H	194±058	28.88±5.66	7.40±3.85	1.12±0.58	4.17±3.12	0.37±0.11
PM	I	209±048	*	*	4.22±0.53	21.73±07.21	*
	II	375±122	*	*	3.52±0.67	30.34±07.35	*
	III	433±097	*	*	3.37±1.10	36.40±11.87	*
	IV+V	448±116	*	*	2.80±0.84	36.66±13.40	*
CV	0-2	*	*	*	4.76±1.79	*	0.28±0.07
	0-3	*	*	*	4.62±1.78	*	0.28±0.05
	0-6	*	*	*	4.97±1.94	*	0.28±0.08
	0-7	*	*	*	5.81±2.45	*	0.31±0.06

* not significant

consistent pattern was detected. There were clearly discernible differences in the copper concentrations of the upper and lower river hard clam populations; the lower river population represented by Stations A, B, C, D and E in general contained less copper than the upper river population (Stations F, G and H) (Table 14). This observation is consistent with relatively high concentrations of copper encountered in the upper river water and sediment samples (Table 6 and 9).

Mean nickel concentrations varied from 5 ppm in C. virginica to 8 ppm in M. mercenaria and P. morrhuana (Table 12). In all three species the highest level of nickel was found in the December 1975 samples. It was also noted that the concentrations of nickel in M. mercenaria gradually increased toward the lower river (Table 14). Such a pattern was contrary to the concentration gradient seen in the water and sediment (Table 6 and 9).

Interspecific differences of cadmium concentrations are clearly evident; C. virginica contains the highest level of cadmium (5.02 ± 1.94 ppm), P. morrhuana with 3.58 ± 1.19 ppm ranks the second and M. mercenaria shows the lowest concentration (1.43 ± 0.58 ppm) (Table 13). In C. virginica and M. mercenaria, again, the highest level of cadmium was detected in the samples obtained on July 1975 when the dredging was completed. The variations of mean cadmium concentrations from 2.90 to 4.30 ppm obtained from various sampling dates in P. morrhuana were considered not significant. However, significant spatial variations in cadmium levels were noted in M. mercenaria and P. morrhuana; the lower river populations showed consistently higher mean levels of

cadmium than the upper river populations (Table 14).

Lead concentrations were at the detection limit of our analytical procedure in 50% of M. mercenaria and 62% of C. virginica samples. Pitar appeared to have a special affinity to lead and attained a mean lead concentration of 31.03 ± 12.25 ppm (N=66) which was 10 to 15 times higher than the concentrations found in M. mercenaria and C. virginica. It was significant to note that in all three species of bivalves the highest level of lead was detected simultaneously in the samples obtained in May 1975 (Table 13), two months before dredging was terminated. During this time the lead content in the surface water was also the highest (Table 3). In the upper river stations the lead concentrations in M. mercenaria and P. morrhuana were again significantly higher (Table 14), while no such concentration gradient that could be correlated with locations was found in C. virginica.

In all three species of shellfish, there were significant temporal variations in mercury concentrations (Table 13). Generally in the pre-and during dredging period the concentrations of mercury were either higher than, or equal to those of the post-dredging months. Mercury concentrations were significantly higher in C. virginica and M. mercenaria samples obtained from upper river stations than those from lower river stations (Table 14).

Analysis of the effect of dredging on trace metals in shellfish. Since predredging data on the concentration of trace metals in the shellfish were limited only to July 1974, it appeared that the best possible way to assess the effect of dredging was to compare the predredging with the post dredging data (July 1974 vs July 1975), and

the during dredging against the post dredging data (November 1974 vs. December 1975; March 1975 vs. March 1976 and May 1975 vs. May 1976). The results of the four paired comparisons using the F test are presented in Table 15, 16 and 17. In C. virginica, only Ni showed significant increase in concentration in tissues collected after dredging, while the concentrations of the other five trace metals (Zn, Cd, Cu, Hg and Pb) decreased significantly (Table 15). It was further noted that the number of trace metals showing decreases increased as the dredging operation proceeded. This apparent correlation might suggest that less trace metals were reaching the oyster seed beds which are located 7-9 nautical miles north of the dredging site.

Of the 14 significant changes in the trace metal concentrations detected in M. mercenaria, 50% of the changes show decreases (Table 16). Specifically cadmium and nickel concentrations were significantly higher in the shellfish during dredging, while the concentrations of copper and mercury decreased noticeably. Zinc and lead concentrations decreased substantially in certain post dredging dates (5/75 vs 5/76, 11/74 vs. 12/75) but increased significantly at other time intervals (7/74 vs. 7/75).

The results of the paired comparisons of trace metals in P. morrhuana are summarized in Table 17. Cadmium and nickel concentrations behaved in the same manner as they did in M. mercenaria. It can be noted that significant reductions of copper and zinc concentrations occurred after dredging in comparison with before dredging as well as during dredging.

When the significant changes of trace metals in the three

TABLE 15. Comparison of mean trace metal concentrations in *Crassostrea virginica* collected on July 1974 and 1975, November 1974 and December 1975, March 1975 and 1976, as well as May 1975 and 1976 from Thames River, Connecticut.

Trace Element	7/74	7/75	F	d.f.	Remarks
Hg	N=4 0.39± 0.02	N=4 0.28± 0.06	24.172*	1,3	D
Zn	11/74 N=4 14213.25±741.94	12/75 N=4 11448.26±3010.03	119.417***	1,3	D
Ni	5.37± 1.08	8.01± 0.91	136.883***	1,3	I
Cd	3.11± 0.52	2.78± 0.44	27.539**	1,3	D
Zn	3/75 N=4 14481.75±2978.15	3/76 N=4 11503.36±4052.08	54.005***	1,3	D
Cu	921.00± 204.40	472.23± 122.53	45.338***	1,3	D
Cd	5.48± 0.78	4.18± 0.57	10.696*	1,3	D
Hg	0.36± 0.02	0.24± 0.02	78.045***	1,3	D
Zn	5/75 N=4 16224.94±3240.11	5/76 N=5 8774.59±2485.05	12.202*	1,3	D
Cu	803.42± 180.63	267.33± 77.02	69.538***	1,3	D
Cd	5.89 0.62	4.02± 0.50	73.704***	1,3	D
Pb	4.16± 2.04	1.25± 1.08	14.974*	1,3	D

*, P < 0.05; **, P < 0.01; ***, P < 0.001. D = Decrease; I = Increase.

TABLE 16. Comparison of mean trace metal concentrations in *Mercenaria mercenaria* collected on July 1974 and 1975, November 1974 and December 1975, March 1975 and 1976, as well as May 1975 and 1976 from Thames River, Connecticut.

Trace Element	7/74	7/75	F	d.f.	Remarks
	N=17	N=14			
Zn	186.88±62.62	285.92±176.83	15.600***	1,19	I
Cd	1.08± 0.24	2.03± 0.29	97.981***	1,19	I
Cu	30.09± 4.74	22.95± 5.97	30.317***	1,19	D
Hg	0.33± 0.14	0.21± 0.09	10.566***	1,19	D
	11/74 N=10	12/75 N=12			
Cu	22.93± 4.90	16.85± 4.69	17.170***	1,10	D
Ni	8.94± 2.48	13.11± 4.39	5.944*	1,10	I
Pb	1.70± 0.00	5.62± 1.42	197.132***	1,10	I
Cd	0.71± 0.35	1.11± 0.60	6.378*	1,10	I
Hg	0.30± 0.12	0.26± 0.11	7.137*	1,10	D
	3/75 N=11	3/76 N=13			
Ni	5.45± 1.87	8.68± 2.35	11.431*	1,12	I
Cd	1.17± 0.31	2.02± 0.28	54.452***	1,12	I
	5/75 N=12	5/76 N=13			
Zn	236.06±46.99	125.14± 27.20	57.530***	1,13	D
Cu	25.25± 3.21	19.47± 6.30	12.112**	1,13	D
Pb	8.01± 3.19	3.24± 1.92	18.232***	1,13	D

*, P < 0.05; **, P < 0.01; ***, P < 0.001; D = Decrease; I = Increase.

TABLE 17. Comparison of trace metal concentrations in *Pitar morrhuana* collected on July 1974 and 1975, November 1974 and December 1975, March 1975 and 1976 as well as May 1975 and 1976.

Trace Element	7/74	7/75	F	d.f.	Remarks
	N=6	N=10			
Pb	18.38 \pm 4.58	36.15 \pm 13.88	21.694***	1,8	I
Cd	3.42 \pm 1.28	4.12 \pm 0.83	5.900*	1,8	I
Cu	20.92 \pm 1.96	15.97 \pm 3.30	8.909*	1,8	D
Hg	0.21 \pm 0.05	0.16 \pm 0.02	6.344*	1,8	D
	11/74	12/75			
	N=9	N=6			
Cu	17.55 \pm 6.51	9.80 \pm 1.95	12.634**	1,7	D
	3/75	3/76			
	N=8	N=11			
Ni	5.15 \pm 0.98	9.25 \pm 1.92	26.743***	1,11	I
Pb	35.67 \pm 9.77	23.01 \pm 5.91	20.612***	1,11	D
	5/75	5/76			
	N=7	N=9			
Pb	45.11 \pm 8.08	23.90 \pm 4.87	36.297***	1,8	D
Zn	389.07 \pm 54.26	226.21 \pm 80.18	54.750***	1,8	D
Hg	0.12 \pm 0.02	0.22 \pm 0.03	38.753***	1,8	I

*, P <0.05; **, P <0.01; ***, P <0.001; D = Decrease; I = Increase

species of shellfish are viewed as a whole, 24 of the 36 cases encountered, or 66%, registered a decrease in the trace metal concentrations.

Gross pathological examinations of the shellfish. A total of 256 C. virginica, 824 M. mercenaria and 660 P. morrhuana were examined for gross pathological conditions from July 1974 to May 1976. No discernible abnormalities were noticed in the inner and outer aspects of gills and palps, as well as the pericardial cavity of the three species of bivalves. In December 1975, six clams were found to contain sediments in their mantle cavity; this was probably due to decreasing water temperatures which reduced the efficiency of the cleansing mechanism of M. mercenaria.

Interspecific variations in the affinity to trace metals shown by the shellfish. There are two lines of evidences in this study, which indicate special affinities of the shellfish for specific trace metals; such evidences are demonstrated by the orders of magnitude of the six trace metals found in the three species of shellfish (Table 18) and the presence of specific interelement correlations as revealed by multiple regression analysis (Table 19, 20 and 21). The predilection for specific trace metals by the shellfish is shown in the following manner: C. virginica appears to be most efficient in taking up zinc, while the relative concentrations of copper, nickel and mercury are the highest in M. mercenaria. P. morrhuana exhibits special affinities for lead and cadmium, since the percent concentrations of these two trace metals in this species are the highest as compared with the other two species.

TABLE 18. Interspecific variations of mean trace element concentrations observed in Crassostrea virginica, (N=33), Mercenaria mercenaria (N=103) and Pitar morrhuana (N=66) from Thames River, Connecticut, July 1974 to May 1976.

Trace Element*	Species		
	<u>Crassostrea virginica</u>	<u>Mercenaria mercenaria</u>	<u>Pitar morrhuana</u>
Zn	13449.51 \pm 4801.90	215.33 \pm 94.43	385.49 \pm 143.52
Cu	754.24 \pm 367.71	23.45 \pm 6.63	16.20 \pm 4.86
Ni	5.52 \pm 1.55	8.55 \pm 3.13	8.06 \pm 2.41
Cd	5.02 \pm 1.94	1.43 \pm 0.58	3.58 \pm 1.19
Pb	2.09 \pm 1.21	3.29 \pm 2.60	31.03 \pm 12.25
Hg	0.29 \pm 0.07	0.25 \pm 0.11	0.19 \pm 0.05

* expressed as mean ppm \pm 1.S.D.

Percent contributions of the six trace metals found in the shellfish

Zn	94.604	85.347	86.715
Cu	5.305	9.294	3.644
Ni	0.039	3.389	1.813
Cd	0.035	0.567	0.805
Pb	0.015	1.304	6.980
Hg	0.002	0.099	0.043
	100.000	100.000	100.000

TABLE 19. Interelement correlations (r) in *Crassostrea virginica* from Thames River, Connecticut, 1974-1976. (N=33).

	Pb	Zn	Cd	Cu	Hg	Mean	S.D.
Ni	.3644*	.2175	-.0959	.3667*	-.0019	5.52	1.55
Pb		.0718	-.0375	-.0584	-.0880	2.09	1.20
Zn			.7290***	.8532***	.4517**	13449.51	4801.90
Cd				.6094***	.5563***	5.02	1.94
Cu					.5600***	754.24	367.71
Hg					---	0.29	0.07

TABLE 20. Interelement correlations (r) in *Mercenaria mercenaria* from Thames River, Connecticut 1974-1976. (N=114).

	Pb	Zn	Cd	Cu	Hg	Mean	S.D.
Ni	.1410	-.0433	.0169	-.3090**	-.1873*	8.59	3.04
Pb		.1432	.1652	.0743	-.0575	3.40	2.79
Zn			.2299*	.2242*	.0065	214.61	93.92
Cd				-.0922	-.4277***	1.43	0.61
Cu					.4749***	23.13	6.91
Hg					---	0.25	0.11

TABLE 21. Interelement correlations (r) in *Pitar morrhuana* from Thames River, Connecticut, 1974-1976. (N=68).

	Pb	Zn	Cd	Cu	Hg	Mean	S.D.
Ni	-.0672	.0652	-.1137	.1911	-.0949	8.09	2.38
Pb		.6086***	-.0267	.1444	-.1194	30.82	12.17
Zn			-.2006	.1614	.0267	357.77	141.97
Cd				-.0712	.0173	3.57	1.18
Cu					-.0988	16.37	4.98
Hg						0.19	0.05

*, P < 0.05; **, P < 0.01; ***, P < 0.001.

In the oysters, significant positive interelement correlations are common among zinc, cadmium, copper and mercury, as well as between nickel and copper. Similar correlations among zinc, cadmium and copper, as well as between copper and mercury are also observed in M. mercenaria; in addition, negative interelement correlations are noted between cadmium and mercury, and among nickel, copper and mercury in this species. In P. morrhuana, however, only one significant interelement correlation is detected, i.e., between lead and zinc; this observation is consistent with the fact that high lead concentrations are detected only in this species of shellfish.

CONCLUSIONS

1. The temperature and salinity regime of Thames River are typical of a partially mixed estuary. Consequently stratifications of temperature, salinity, and oxygen and chlorophyll are most prominent in the summer and least obvious in the winter.
2. Effects of dredging on the phytoplankton production are probably minimal as demonstrated by the sediment elutriate experiments.
3. Two field studies conducted on April 9 and May 14, 1975 suggest that the effect of dredging on phytoplankton production was transient and limited to within 500 yds of the dredge-barge. Furthermore, the concentrations of mercury in the area traversed by the plume, were consistently lower than that of the background.
4. Mean concentrations of the six trace metals in water samples from Thames River are: zinc, 11-28 ppb; nickel, 3-10 ppb; copper and lead, 1-5 ppb; cadmium, less than 1 ppb; and mercury, 4-50 parts per trillion.

5. The concentrations of the six trace metals in the water column vary significantly with dates and transects. The highest concentrations of Ni, Pb, Cd and Hg were observed either before or during dredging. Copper concentrations on the other hand, were significantly higher in the post dredging period. Higher Ni, Pb and Cu concentrations are most frequently associated with the upper river water samples, while Hg levels in the surface water are significantly higher in the lower river transects.
6. The concentrations of Ni, Cu and Pb are significantly influenced by the tide. During the low tide, significant differences in copper concentrations are noted in the surface and bottom water samples.
7. Mercury concentrations are correlated with ambient water temperature, while copper concentrations are negatively correlated with salinities.
8. The concentrations of the six trace metals and percent organic carbon in the channel surficial sediments tend to increase in an upriver direction; areas of high concentrations are generally associated with foci of industrial activities along the river. In comparing the predredging with post-dredging data, there is a noticeable post-dredging reduction of zinc ($P < 0.005$) and copper ($P < 0.05$) concentrations in the lower river sediment.
9. With the exception of nickel, the concentration of the remaining trace metals is significantly correlated with percent organic carbon contents of the sediment, suggesting that the percent organic carbon is a good predictor for Zn, Pb, Cu, Cd and Hg concentrations in the sediment.

10. Although nearly all the analysis of variance results suggest that the concentrations of the six trace metals in the three species, of shellfish sampled at different times are statistically significant, it is nevertheless, difficult to separate the normal seasonal variation in trace metal body burden from the change attributable to dredging. However, in C. virginica, with the exception of Ni, the overwhelmingly significant decrease in Zn, Cd, Cu, Pb and Hg concentrations probably reflects the general reduction of the trace metals in the environment, particularly in the lower river. It is more difficult to interpret the M. mercenaria results (Table 16) in which the number of trace metals showing increases in their concentrations equals that of decreases. The fact that copper and mercury consistently exhibit lower concentrations in post dredging samples, may be indicative of a cleaner environment.

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C. Physical Oceanography of Disposal Area
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INTRODUCTION

Two important physical characteristics of marine dredge spoil disposal sites are the movement of water in the disposal area (the general circulation), and the optical properties of the water mass(es) occupying the area. In general, disposed dredge spoil will not significantly affect the general circulation of the disposal area provided, of course, that the ratio of the volume of the disposal area to the volume of spoil material remains large. However, the general circulation may have very significant effects upon the fate of the disposed spoil material both at the time of disposal through direct transport of the material and through a long time effect due to erosion of the deposited spoil. Optical properties of the water will be affected by the disposal process through what are generally large increases in turbidity at the time of disposal, and to a less significant extent, through resuspension or erosion of the deposited material in time.

A two year study was undertaken under a contract with the Sandy Hook Laboratory, Northeast Fisheries Center, of the National Marine Fisheries Service under the National Oceanic & Atmospheric Administration, U.S. Department of Commerce, to study the general circulation of the New London Dump Site by Eulerian and Lagrangian techniques to determine the transport of the disposed material. Light transmissivity was also measured at the

Dump Site and the surrounding area. This report is a summary of that study.

Bohlen (1975) has shown that the concentration of suspended materials in the eastern end of Long Island Sound is a complex function of currents, river discharges, meteorological effects, biological activity, and morphology. The concentration of suspended material is complicated even further by the disposal of dredge spoil material. In general, the experiments performed in the study of the New London Dump Site Area were designed with the aim of acquiring knowledge of the general circulation of the area and background as well as the anomalous turbidity in the water as a result of active disposal. Here, we are using turbidity as a resultant effect of suspended materials.

A map showing the dump site, sampling stations, and surrounding area is shown in Figure 1. Actual sampling has been on a seasonal basis.

Preliminary studies of the area have been carried out by the Physical Oceanography Division of the U.S. Naval Oceanographic Office (NAVOCEANO, 1973), and a concurrent study by Morton *et al.* (1975) for which only a preliminary summary was available for review.

METHODS AND INSTRUMENTATION

Station locations are shown in Figure 1 and tabulated in Table 1. Actual programs for station sampling varied and have been described in previous reports or will be discussed in the results to follow.

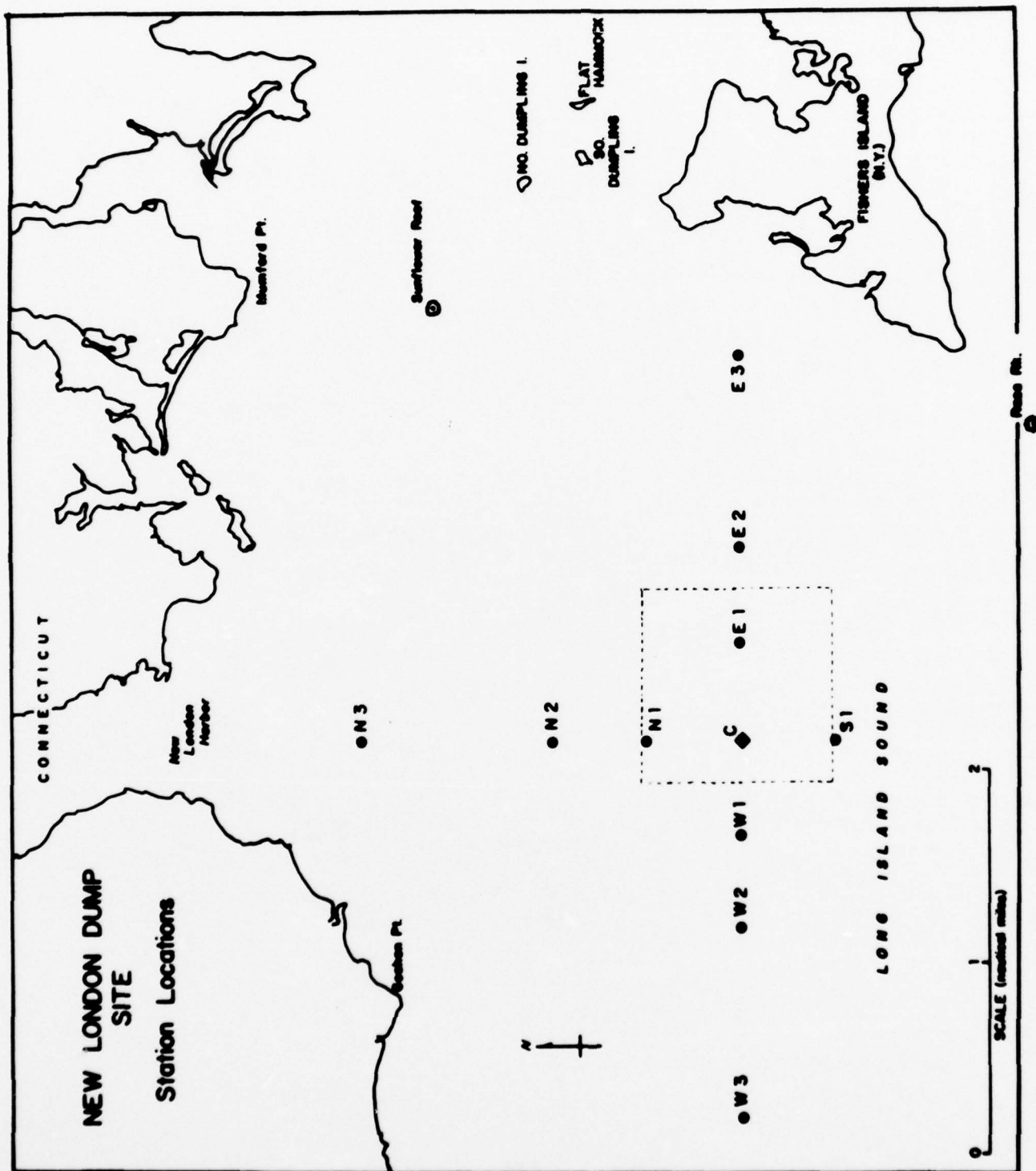


Figure 1: Map of the dump site and station locations. "C" is the Center Buoy.

Table 1. Station locations in the area of the New London Dump Site.

Station	Latitude	Longitude
Center Buoy	41°16'08"	72°05'00"
W1	"	72°05'40"
W2	"	72°06'20"
W3	"	72°07'37"
E1	"	72°04'20"
E2	"	72°03'37"
E3	"	72°02'17"
S1	41°15'38"	72°05'00"
N1	41°16'38"	"
N2	41°17'08"	"
N3	41°18'08"	"
NW1	41°17'14"	72°06'08"
NE1	41°16'50"	72°04'04"

Vertical temperature profiles were obtained using bathythermographs and in some cases Beckman Instrument RS-5 Temperature-Conductivity Probes. In either case, surface temperatures were measured with standard mercury thermometers for reference.

Salinities were obtained from drawn water samples and analyzed by the New York Ocean Science Laboratory Chemical Oceanography group, or, in some cases, obtained through *in situ* conductivity measurements using the Beckman Instrument RS-5 Temperature-Conductivity Probe.

The current meters used in this study were General Oceanics Model 2010 film recording meters. A 3 meter subsurface array was generally implanted close to the Center Buoy of the dump site (Figure 1), and on occasion, single bottom meters surrounding a spoil mound or at other stations. The film from the meters was processed in the laboratory's photo-lab and

read frame by frame using a standard micro-film reader, recording date, time, direction, and speed from each frame. These data were then processed in a computer program whose output includes the average speed over 15 minute intervals along with the average north-south (v) and east-west (u) components of the current. At the end of each half-tidal cycle in the data, the program output then includes the average u and v components over the half-tidal cycle, the resultant speed (R), direction (θ), the effective distance traveled (D), and Reynolds stresses.

The 15-minute average serves as a very effective low-pass filter thus reducing the effects of surface waves in the current data.

Beam transmittance measurements were obtained with a HYDRO-Products Transmissometer equipped with a depth sensor. Normally, the transmissometer is nulled or standardized in air at a setting of 92% that allows for reflection, approximately 4% (HYDRO-Products, personal communication), that takes place at the two glass/air interfaces of the transmissometer system. When the transmissometer is then operated in water, the reflective losses at the glass/water interfaces are much less than they were at the glass/air interfaces by approximately an order of magnitude, hence, it is possible for the transmissometer to show higher values of transmissivity in clear water than the null position of 92% in air. For an estimated error in transmissivity of 1%, the relative error in the beam attenuation coefficient is 10% at 90% transmissivity.

DEFINITIONS

The following definitions are for the terms and coefficients used in this report. Other terms used in previous reports are not necessarily covered herein, except as they appear in this report.

a. Currents:

u: The East/West velocity component in cm/sec.

v: The North/South velocity component in cm/sec.

NOTE: The above velocity components are readily computed from measured current speed, R, and direction relative to geographic north, θ , by

$$u = R \cos \alpha$$

$$v = R \sin \alpha$$

where

$$\alpha = 90^\circ - \theta \text{ for } 0^\circ \leq \theta \leq 90^\circ$$

or
$$\alpha = 450^\circ - \theta \text{ for } 90^\circ < \theta < 360^\circ.$$

b. Beam Attenuation Coefficient β : The sum of the absorption coefficient and total scattering coefficient and calculated from

$$\beta = -(1/L) \ln (T\%/100)$$

where T is the beam transmittance in percent and measured with the transmissometer and L is the path length in meters so that the units of β are per meter.

RESULTS

1. Temperature and salinity

Results from individual cruises have been presented in previously submitted reports so that this report represents a general summary of past results, including the observations from the last cruise (8 and 9 June 1976) that were not covered in a previous report. Individual cruise temperature and salinity data for surface and bottom have been averaged by month for the various stations and are shown in Tables 2a and b, respectively.

In general, the stations are out of the direct influence of the Thames River; the possible exception is Station N3 at the mouth of the harbor (Fig. 1) where occasional excursions of less saline and often warmer waters from the river have been observed. Ensemble averages were therefore calculated and appear in the last column in Table 2a and b. For comparison, the ensemble data are plotted against similar data from the National Ocean Survey Tide Gage Station (NOS Publ. No. 31-1, 1972) on Plum Island (approximately 10 miles south and west of New London Harbor) in Figure 2. As can be seen in this figure, the results from the New London Dump Site Area fall within the ranges specified by the vertical bars representing the averaged maximum and minimum values for Plum Island.

The annual range in salinity is small, generally less than 3‰ except at Station N3 at the mouth of the harbor where the range was close to 10‰ . Temperatures for the area range between 0 and 20 degrees Celsius during the year.

Table 2a. Monthly mean surface and bottom temperatures, New London Dump Site Stations

Surface Temperature (°C)												
Month	CB	N3	N2	N1	S1	E3	E2	E1	W1	W2	W3	Average
February	2.9	-	-	0.7	1.8	-	-	1.2	3.0	-	-	1.9
March	3.7	-	-	-	-	-	-	-	4.0	3.9	3.4	3.8
May	10.0	15.1	9.8	12.9	10.2	9.9	10.0	10.4	11.1	10.0	10.0	10.8
June	12.6	14.3	12.2	12.1	12.1	12.3	12.9	12.7	12.5	12.4	12.5	12.6
July	19.0	19.4	18.5	19.5	19.7	18.8	18.7	19.5	19.5	18.4	18.7	19.1
August	19.6	19.2	19.6	19.6	19.7	19.6	19.6	19.6	19.9	19.8	19.3	19.6
September	19.3	-	-	-	-	18.7	18.7	20.8	20.2	20.0	20.4	19.7
December	9.4	8.5	7.9	10.5	9.2	8.1	7.4	9.0	9.1	7.8	9.2	8.7

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Bottom Temperature (°C)												
Month	CB	N3	N2	N1	S1	E3	E2	E1	W1	W2	W3	Average
February	3.1	-	-	1.1	2.0	-	-	2.4	3.0	-	-	2.3
March	3.8	-	-	-	-	-	-	-	4.0	4.0	3.7	3.9
May	9.2	11.8	9.3	10.8	9.6	9.8	9.5	9.3	10.0	10.0	9.6	9.9
June	11.9	12.2	12.1	12.0	11.4	12.1	11.9	11.9	11.9	11.9	12.0	11.9
July	17.9	17.1	17.8	18.5	18.6	17.2	16.9	18.6	18.6	17.7	17.8	17.9
August	18.7	18.9	19.0	18.9	18.0	19.3	19.2	18.3	18.8	18.9	18.2	18.7
September	18.9	-	-	-	-	18.4	18.2	20.3	20.0	20.0	19.8	19.4
December	10.8	8.8	7.6	10.8	9.5	8.4	7.8	9.2	9.4	8.0	9.2	9.0

Table 2b: Monthly mean surface and bottom salinities, New London Dump Site Stations

Surface Salinity (‰)												
Month	CB	N3	N2	N1	S1	E3	E2	E1	W1	W2	W3	Average
February	29.85	-	-	28.56	28.37	-	-	25.70	29.07	-	-	28.31
March	29.82	-	-	-	-	-	-	-	29.46	29.40	28.52	29.30
May	29.58	20.04	29.55	26.36	29.41	29.56	29.47	29.45	29.09	29.72	29.54	28.34
June												
July	30.15	30.11	30.24	30.24	29.78	30.15	30.07	30.14	30.18	30.26	30.05	30.12
August	30.62	30.03	30.42	31.29	30.34	31.05	30.89	30.96	29.60	30.56	30.37	30.56
September	30.48	-	-	-	-	30.51	29.78	29.59	30.05	30.24	30.36	30.14
December	30.25	24.17	30.59	29.59	29.71	25.72	28.87	29.86	30.58	31.34	30.48	29.20

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Bottom Salinity (‰)												
Month	CB	N3	N2	N1	S1	E3	E2	E1	W1	W2	W3	Average
February	30.30	-	-	29.66	29.69	-	-	30.42	30.02	-	-	30.02
March	30.45	-	-	-	-	-	-	-	30.17	30.40	30.15	30.29
May	29.90	29.75	29.78	29.84	29.88	29.71	29.81	29.82	29.97	30.52	29.70	29.86
June												
July	30.53	30.41	30.48	30.46	30.43	30.41	30.44	30.45	30.14	30.52	30.49	30.43
August	31.45	30.94	31.02	31.22	31.37	31.46	31.43	31.94	30.92	30.62	31.15	31.23
September	30.84	-	-	-	-	30.56	30.60	30.61	30.53	30.60	30.81	30.65
December	30.67	30.13	31.19	29.84	30.68	30.15	30.92	30.48	30.84	31.80	30.72	30.67

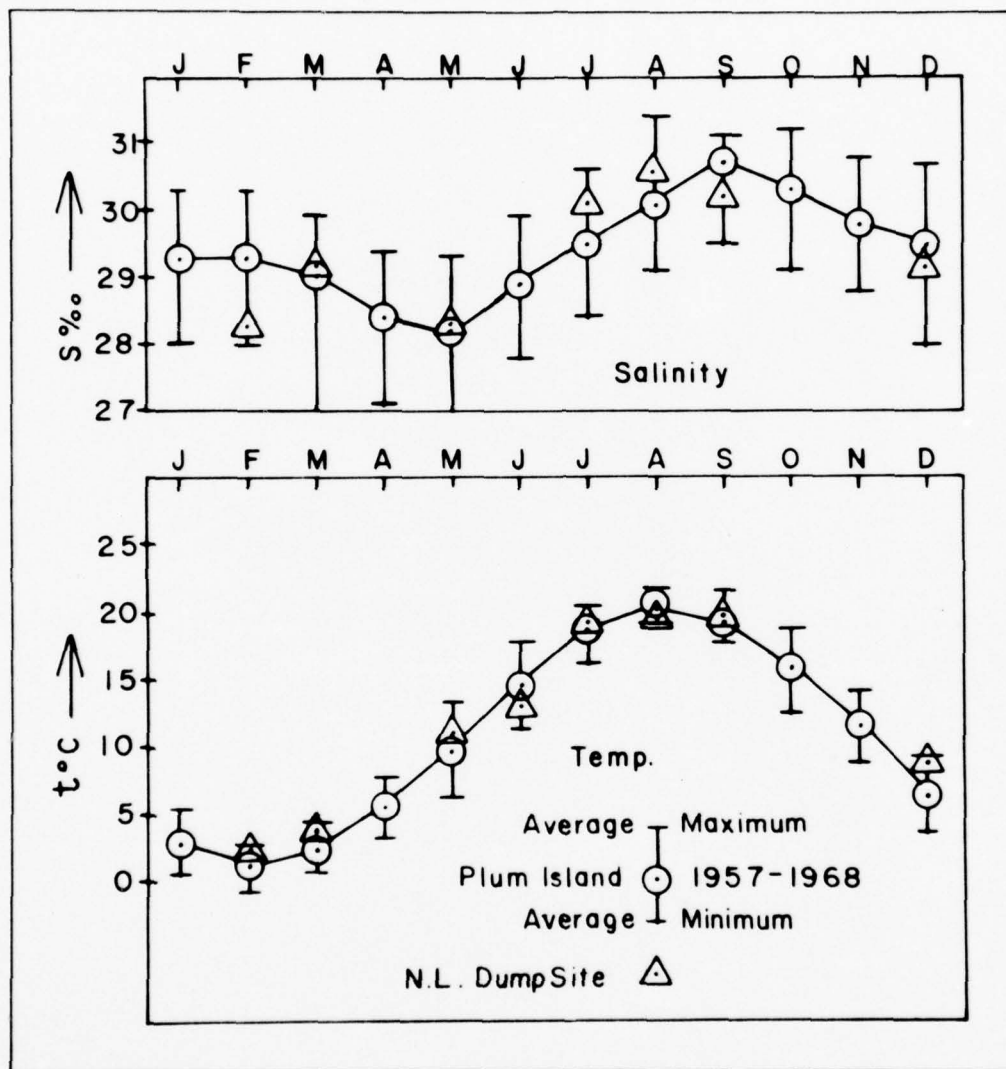


Figure 2: Monthly average temperatures and salinities relative to 15°C as calculated from National Ocean Survey Tide Gage Station Data for Plum Island, N.Y. (NOS Publ. No. 31-1, 1972), with the mean monthly maximum and minimum values and the ensemble average temperatures and salinities from the New London Dump Site Area.

Vertical variations in both salinity and temperature are also small; temperature gradients were approximately 2°C and salinity gradients were approximately $1^{\circ}/\text{‰}$, except at N3 where gradients of 5°C and $10^{\circ}/\text{‰}$ were observed.

In summary, the results obtained in the New London Dump Site Area are within expected temperature and salinity limits when compared to historical data from National Ocean Survey and when compared to a 3 year data base for Eastern Long Island and Block Island Sounds (Hollman, 1976).

2. Transmissivity

Individual cruise results for the attenuation coefficient, β , were ensemble averaged (spatially composited) and used to calculate monthly averages representative of the entire dump site area. The data include all pre-dump and post-dump results but do not include results obtained during and immediately after a dump. In effect, the data are "background" and are tabulated in Table 3, and plotted by month in Figure 3. As shown in the figure, maximum β values and hence minimum background transmissivity, occurs in summer and minimum β , hence maximum transmissivity during winter months. As is also evident from the table and figure, β increases with depth, that is, in the average, transmissivity is lower in the bottom layer than in the surface layer. The annual range of β is 1.45m^{-1} at the surface, 1.53m^{-1} at mid-depth, and 0.94m^{-1} at the bottom.

Table 3. Average monthly beam attenuation coefficients calculated from ensemble averages, New London Dump Site area.

Month	$\beta (m^{-1})$		
	Surface	Mid-depth	Bottom
Feb.	0.78	1.03	1.30
Mar.	0.41	0.41	1.28
Jun.	1.14	1.14	1.19
Jul.	1.46	1.84	2.06
Aug.	1.17	1.45	1.80
Sep.	1.86	1.94	1.95
Oct.	1.20	1.18	1.22
Dec.	1.01	1.04	1.12
Avg.	1.13	1.25	1.49

Surface: 1m
 Mid-depth: 9.7m
 Bottom 18.7m

The transmissometer was also towed through the area while continuously recording to obtain estimates of spatial variability (see Quarterly Report for July through September, 1975). These tows were made along the East/West and North/South transects (Fig.1). Average results are tabulated in Table 4. There was no indication in the East/West distribution of β values that there was a concentration of spoil material by Center Buoy station. The distributions were either very irregular within the limits indicated in Table 4, or, in one case, showed a very slight tendency to decrease in the upstream direction away from the Center Buoy station.

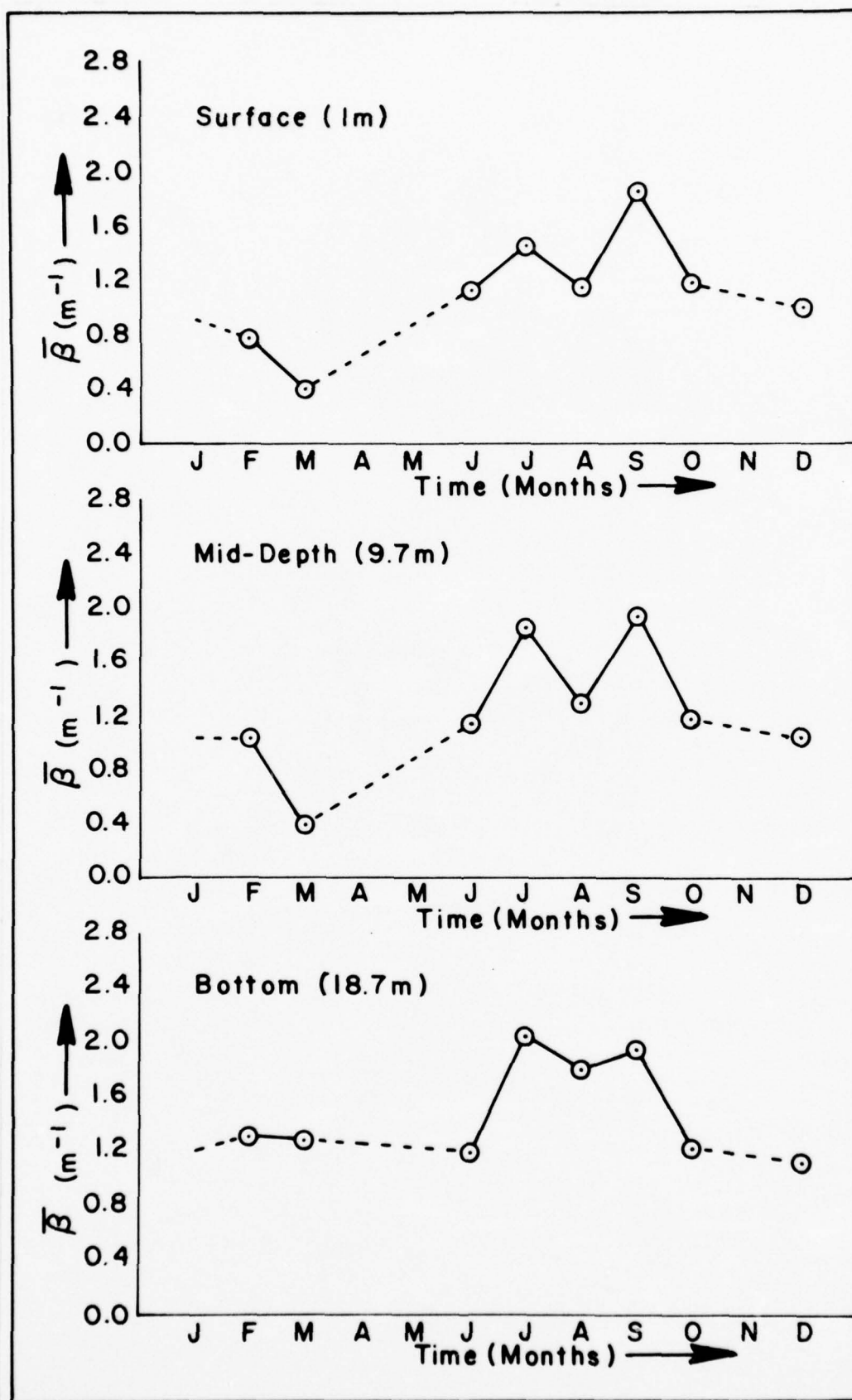


Figure 3: Mean monthly beam attenuation calculated from ensemble averages for surface, mid-depth, and bottom layers, New London Dump Site Area.

For the North/South tows, the tendency was for β to decrease (clearer water) toward the south and Station Center Buoy. Again, the average values for β in Table 4 are in keeping with the annual trend with highest values occurring in August and decreasing toward winter. No significant difference can be found between the East/West and North/South tows. The highest coefficient of variation (standard deviation divided by the mean) is 13% during August on the East/West transect. The average range for these spatial variations was 0.43m^{-1} , with a maximum value of 0.89m^{-1} at the East/West transect in August and a minimum of 0.13m^{-1} on the North/South transect in April.

Table 4. Average beam attenuation coefficient, $\bar{\beta}$, with standard deviation and range calculated from continuous measurements made while towing the transmissometer along East/West and North/South transects.

Time	Mean Depth	East/West			North/South		
		$\beta(\text{m}^{-1})$	S.D.*	Range**	$\beta(\text{m}^{-1})$	S.D.	Range
Aug 1975	7.8m	1.43	0.19	0.89	1.43	0.12	0.42
Apr 1976	5.5m	0.98	0.11	0.41	0.85	0.04	0.13
Jun 1976	6.4m	1.26	0.08	0.52	1.21	0.05	0.23

*S.D.: Standard Deviation; **Range: Maximum value observed minus minimum value.

A number of experiments were performed during dumping operations before their cessation July 1975. One such experiment was to anchor a vessel downstream of the dump site and monitor the turbidity cloud with the transmissometer as the cloud passed the anchored vessel. The results from

these experiments are tabulated in Table 5. The elapsed times required for the site to return to background were approximately 48 minutes for the downstream cases, and 29 minutes for the upstream case. In all experiments, there were at least 2 distinct clouds that passed the stations (see Figure 4); in the upstream example, there were 3. These clouds of material are approximately 10 minutes apart; the second cloud appears to be longer in duration than the initial one. Gordon (1974) showed similar features for disposal experiments in western Long Island Sound. In all cases, the range in β was from infinite values to approximately 1 as background. In Figure 4, the ordinate for the July 9th experiment is not transmissivity, $T\%$, but $\tau(\%)$ which was the observed transmissivity normalized to background values. The transmissometer developed a weak power supply (but stable over the time period), so that the absolute transmissivity values were unreliable. However, the normalized or relative values are useful for studying the distribution of the turbidity cloud in time.

Table 5. Elapsed time required for the dump site area to return to background transmissivity values following a barge release.

Date	Height off Bottom	Distance from Barge	Elapsed Time	Mean Current
20 May 1975	1m	100m downstream	49 min	22 cm/sec, Ebb
21 May 1975	2.2m	50m downstream	29 min	24 cm/sec, Ebb
9 Jul 1975	1m	80m downstream	48 min	41 cm/sec, Ebb

Vertical transmissometer profiles were made while tracking drogues planted

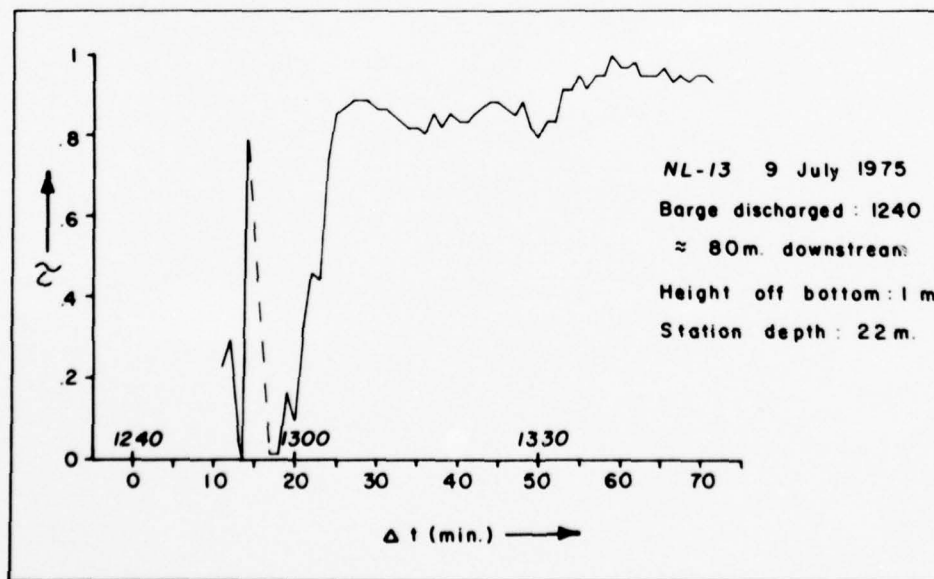
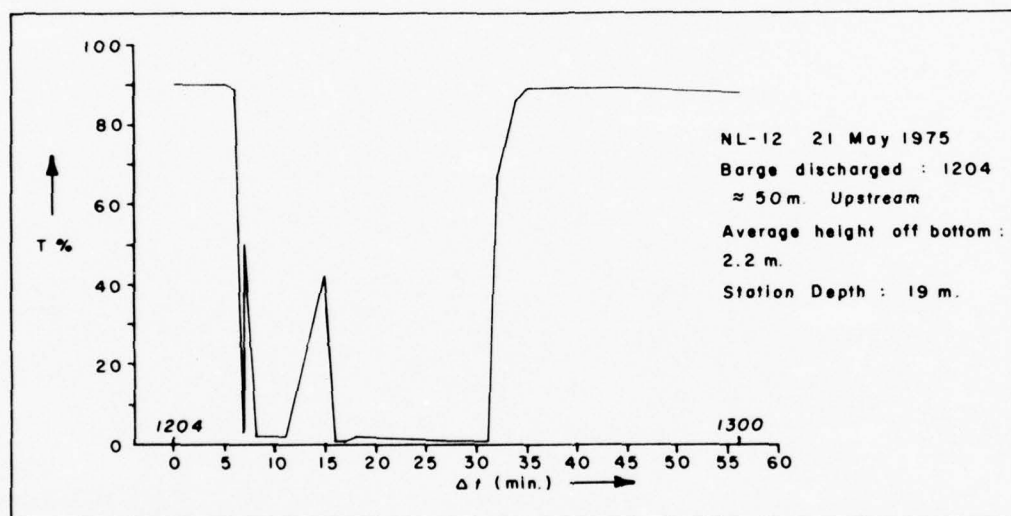
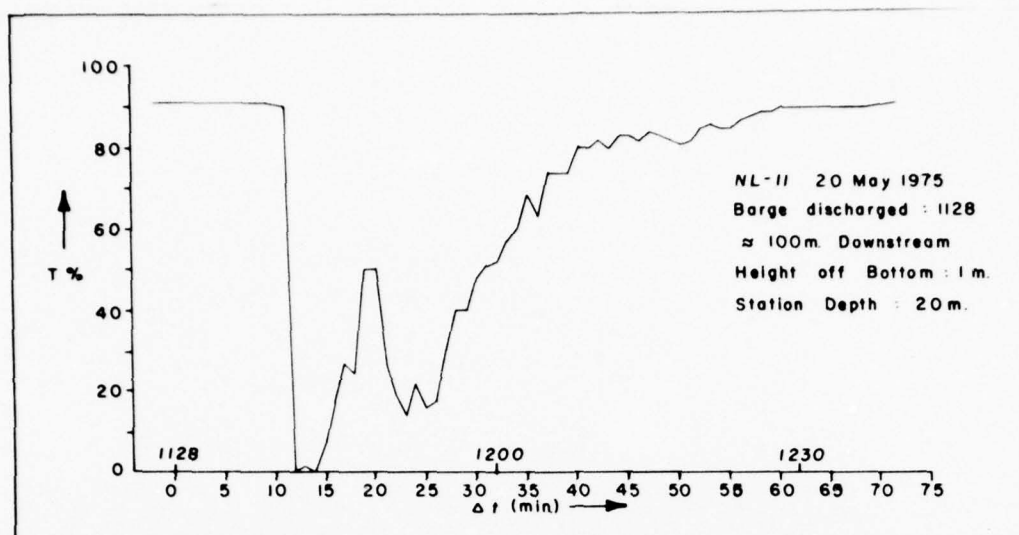


Figure 4: Continuous transmissivity values as a function of time at fixed positions downstream and upstream of the disposal point.

in the wake of a discharging barge. The results of an extreme case are tabulated in Table 6. Other experiments did not pick up a detectable cloud, or plume, relative to background in either the surface or bottom layer within 5 to 10 minutes of the barge release; in these cases the tide was ebbing near maximum strength. In the extreme case shown in Table 6, the current was almost slack, the mean velocity was 9.3 cm/sec over the 65 minute time span calculated from the current meter data. The total distance that the drogue traveled was only 550 meters based on fixes obtained by visual bearings. In the previous case (Table 5) under mean flow conditions, the time required for the area to return to background was approximately 48 minutes; in this case, staying with the plume under near slack flow, the time required to return to background was 65 minutes. In the experiments carried out near maximum flow conditions, the plume as identified by transmissivity values virtually disappeared within 10 to 15 minutes of the release.

Table 6. Elapsed time required for a plume to return to background as measured by tracking a bottom drogue planted in the wake of a discharging barge at the New London Dump Site.

Date	Drogue	Depth	Distance	Elapsed Time	Mean Current
11 Sep 1975	Bottom	20m	550m	65 min	9.3 cm/sec, Ebb

Continuous transmissometer measurements were made over a period of 1-1/2 hours, 1m off the bottom, approximately 230m downstream of a spoil mound on July 17, 1975. During the sampling period, bottom turbidity values in-

creased from 78% at the start to 82% near the mid-point of the period (close to the time of maximum ebb at The Race) and decreased again to 80% at the end of the observation period. The average of all values was 80.1% ($\beta=2.22\text{m}^{-1}$) with a standard deviation of 1% (0.12m^{-1} for β). Upstream (228m) transmissometer measurements were obtained within 29 minutes of the last downstream bottom reading. The results of the last vertical profile downstream and the upstream station are tabulated in Table 7. For comparison, the maximum observed variation in β over 29 minute periods was at most 5%, in fact, the coefficient of variation (ratio of standard deviation to the mean) was 5.4% for the entire 1-1/2 hour recording period. The maximum observed bottom flow as measured on top of the spoil mound was approximately 1 knot, and occurred shortly before the transmissometer recording began; maximum observed flow during the period of the experiment was 41 cm/sec at the beginning; the mean speed was 33 cm/sec.

Table 7. Transmissivity (T%) and attenuation coefficient (β) taken approximately 228m upstream and downstream of a spoil mound, New London Dump Site, 17 July 1975. The transmissometer path length was 0.1m.

Downstream			Upstream		
Depth	T (%)	$\beta(\text{m}^{-1})$	Depth	T (%)	$\beta(\text{m}^{-1})$
1m	83	1.86	2m	84	1.74
6m	83	1.86	6m	83	1.86
11m	82	1.98	11m	83	1.86
16m	81	2.11	16m	82	1.98
20m	80	2.23	18m	82	1.98

This experiment of July 17th was followed by another scouring study on 23 October where two stations, one 280m downstream of a spoil mound and the other 240m upstream, were sampled alternately over the period of maximum flood in the morning and again during the ebb cycle in the afternoon. The results are tabulated in Tables 8a and b, respectively. As can be seen in the table, bottom transmissivity values increased downstream (West station during flood) by 5%; relative increases in β ranged from 8% to 15%. During the ebb, bottom transmissivity again increased by 5% in the downstream direction (East station during ebb) with a relative increase in β of 12%.

Average downstream transmissivities (West station) during the flood tide were 28% and 26% (using a 1 meter path length) over two separate 5 minute continuous sampling periods. The mean range was 6%. These average percentages yield values for β of 1.27m^{-1} and 1.35m^{-1} respectively. At the downstream station (eastern) the two 5 minute averages were 30 and 31% with a mean range of 4%, corresponding values for β were 1.20m^{-1} and 1.17m^{-1} respectively. Similarly, during the ebb cycle later in the day, the downstream average was 32% ($\beta = 1.14\text{m}^{-1}$) with a maximum range of 4% (East station during ebb) and the upstream average was 35% ($\beta = 1.05\text{m}^{-1}$) with a range of only 1%. In all 5 minute sampling periods, the transmissometer was approximately 1 meter off the bottom. The average flow during the flood sampling period (0930 to 1011) was approximately 17 cm/sec and approximately 40 cm/sec during the ebb in the afternoon.

Table 8a. Percent beam transmittance (T) for 1 meter path length and attenuation coefficient (β) as a function of depth for the positions 240m west and 280m east of the spoil mound, 23 October 1975, during flood tide.

Station		west		east		west		east	
Time		0930		0947		1000		1011	
Depth (m)	T(%)	β (1/m)	Depth (m)	T(%)	β (1/m)	Depth (m)	T(%)	β (1/m)	T(%)
1	31	1.17	1	30	1.20	1	31	1.17	30
5	31	1.17	5	32	1.14	5	32	1.14	32
10	29	1.24	10	32	1.14	10	33	1.11	29
15	29	1.24	15	30	1.20	15	30	1.20	29
20	27	1.31	17	30	1.20	20	25	1.39	29
									1.24
									1.24

Table 8b. Percent beam transmittance (T) for 1 meter path length and attenuation coefficient (β) as a function of depth for the positions 240m west and 280m east of the spoil mound, 23 October 1975, during ebb tide.

Station	east			west		
Time	1416			1517		
Depth (m)	T(%)	β (1/m)	Depth (m)	T(%)	β (1/m)	
1	28	1.27	1	31	1.17	
5	29	1.24	5	31	1.17	
10	32	1.14	10	36	1.02	
15	31	1.17	15	39	0.94	
17	30	1.20	20	35	1.05	

3. Currents

a. Eulerian Current Measurements

Most observations of the currents in the New London Dump Site have been made from 3 meter sub-surface arrays moored close to the Center Buoy (Fig. 1). These results have been presented graphically as East/West (u) and North/South (v) components of the flow averaged over 15 minutes in past quarterly reports.

December 11, 1974.

In order to standardize the results, interpolations in time of the East/West and North/South velocity components were made for the respective values at the time of high water at New London and at -6 hours to +6 hours around this time. The time of high water was obtained from the "Tide Tables, High & Low Water Predictions, East Coast of North and South America" published by the U.S. Department of Commerce, National Ocean Survey/NOAA for the years 1974 through 1976. These hourly velocity components relative to the time of high water (EST) at New London were then time averaged and in effect filtered seasonal differences, diurnal inequalities, and wind effects. The result is an estimate of the average half-lunar day currents at the dump site. These averages are tabulated in Table 9, and shown in Figure 5 for surface (4m), mid-depth (10m), and bottom flow (17m).

Table 9. Average hourly East/West (\bar{u}) and North/South (\bar{v}) velocities relative to the time of High Water at New London.

Time	Surface (4m)		Mid-depth (10m)		Bottom (17m)	
	\bar{u}	\bar{v}	\bar{u}	\bar{v}	\bar{u}	\bar{v}
-5 hours	23.1	-7.0	12.9	-0.4	4.8	4.2
-4 "	5.1	-3.7	-7.1	2.2	-16.9	7.6
-3 "	-17.8	2.1	-34.9	15.0	-34.2	12.7
-2 "	-21.5	12.6	-32.5	26.6	-28.4	13.4
-1 hour	-9.7	7.5	-3.9	15.2	-9.0	8.9
High Water	0.6	-8.5	7.7	-3.2	2.4	2.4
+1 hour	7.8	-20.9	19.2	-16.5	13.6	-3.1
+2 hours	18.6	-21.3	34.7	-24.8	31.2	-10.7
+3 "	48.1	-15.2	54.8	-18.6	38.1	-10.0
+4 "	64.1	1.6	59.5	-1.1	36.2	-6.7
+5 "	56.6	-5.1	55.4	2.7	31.0	2.6
+6 "	37.8	-9.8	43.7	-1.2	24.3	3.5
Average	17.7	-5.6	17.4	-0.3	7.8	2.1

As can be noted in Figure 5, the v -component of the velocity changes direction (sign) from south to north (positive) at hour 4 during the ebb cycle. This feature is detectable in nearly all of the current meter observations, in some cases more dramatically so than in others. This means that 4 hours after the time of high water at New London, the ebb flow changes direction from the southeast quadrant to the northeast

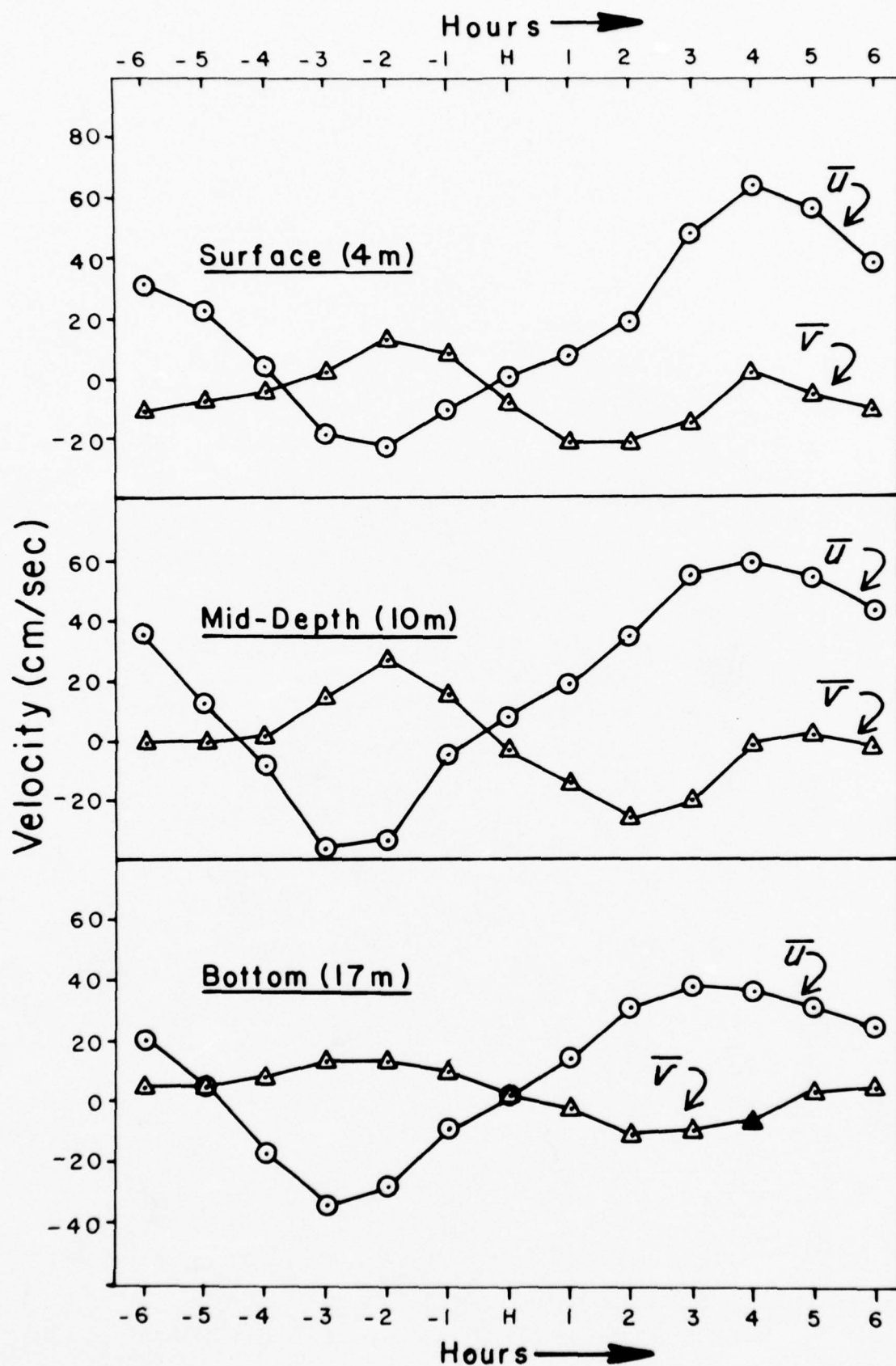


Figure 5: Average hourly velocity components centered around the time of Highwater, H, at New London for the surface (4m), mid-depth (10m), and bottom (17m). The East/West component is \bar{u} and the North/South component is \bar{v} .

quadrant. In the bottom layer, this change in direction occurs at hour 5. From these averages, the time of slack water and ebb flow begins, is close to the time of high water, H; slack, flood begins is approximately 4 hours earlier than high water (-4 hours) at the surface and mid-depth, and 5 hours earlier at the bottom. The duration of the flood is also greater by approximately 1 hour in the bottom layer. Maximum flood velocities occur approximately 3 hours earlier than high water at the bottom, and 2 hours earlier at the surface and mid-depth. Maximum ebb occurs 4 hours after high water at both the surface and mid-depth and 3 hours after at the bottom.

The average velocity components in Table 9 have also been plotted as hourly vectors from a common origin as shown in Figure 6. Resultant vectors for each depth appear directly beneath their corresponding 12 hourly vectors. North is parallel to the border toward the top of the page. Note the short duration of the flood cycle; for the surface, the flood flow involves only hours -3 to -1, and -4 to -1 at mid-depth and bottom. The magnitude of the flood is also greater at mid-depth and bottom than at the surface. The resultant flow, that is, the net flow, is SE for the surface, E at mid-depth, and NE at the bottom; a cyclonic rotation with depth.

Maximum speeds based on 15 min averages have been tabulated in Table 10. It can be seen that the highest speeds occur during the ebb in the surface layer as would be expected; the values range from a maximum

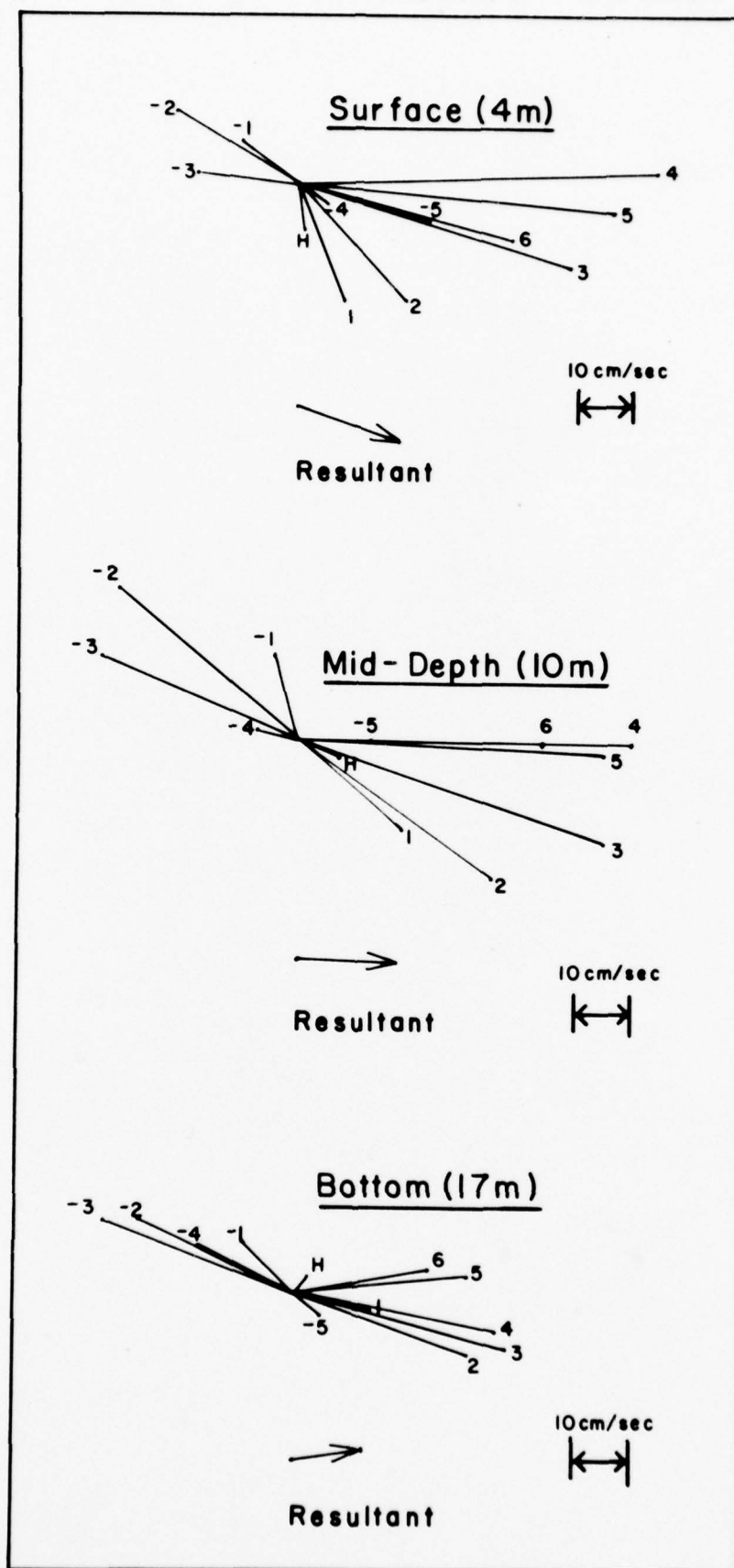


Figure 6: Polar diagram of average hourly velocities relative to the time of High water, H, at New London. Geographic or true North is parallel to the border and toward the top of the figure. The resultant vector flow for each layer is directly beneath the respective polar diagram.

Table 10. Maximum observed velocities over 15 minute averages in cm/sec-1.

Date	Surface		Mid-depth		Bottom	
	Flood	Ebb	Flood	Ebb	Flood	Ebb
5 Aug 74	51.4	77.6	59.0	74.0	47.5	47.5
11 Sep 74	25.1	73.5*	41.3	46.1*	30.0	28.3*
11 Dec 74	-	86.4*	-	80.2*	-	51.2*
28 Feb 75	-	100.0	-	59.3	-	61.1
6 Mar 75	-	70.8	-	49.0	-	33.6
20 May 75	49.1	92.0	0	0	41.8	50.3
21 May 75	56.4	-	-	-	40.5	56.6
9 Jul 75	-	-	-	-	-	45.1
17 Jul 75	-	-	-	-	-	51.5*
22 Jul 75	-	76.5	-	66.5	-	48.9
25 Aug 75	37.1	77.5	44.3	71.9	33.1	44.0
26 Aug 75	25.0	70.7	40.2	66.7	38.0	41.9
16 Sep 75	19.3*	65.9*	36.4*	75.1*	37.1*	48.8*
8 Oct 75	-	81.5	-	64.9	-	59.4
23 Oct 75	-	-	-	-	-	45.8**
4 Dec 75	52.3	65.4	52.6	69.4	49.4	56.2
5 Dec 75	51.3	60.8	51.9	61.7	45.3	54.9
9 Feb 76	-	41.5	-	-	36.4	-
10 Feb 76	31.2	82.3	-	-	34.3	37.4
11 Feb 76	43.1	58.4	-	-	43.9	34.0
12 Feb 76	44.0	82.8	-	-	40.2	33.5

* Incomplete tidal cycle - not representing maximum tidal flow

** Average of 3 bottom mounted meters in dump site

of 100 cm/sec to a minimum of 41.5 cm/sec. It should be borne in mind that tidal currents in The Race can vary as much as 48% in one month due to spring and neap tides, and as much as 57% over the period of a year. Therefore, the relative range of 58.5% is not out of line considering also the added influence of winds. The range for the flood in the surface layer ranges from a maximum of 52.3 cm/sec to a minimum of 19.3 cm/sec, although the latter was not part of a complete flood cycle. Lowest speeds were encountered in the bottom layer, also as would be expected. In this layer, the difference between flood and ebb speeds are much less as compared to the differences observed in the surface and there are a few instances where the maximum flood speed was greater than the corresponding maximum ebb speed. The average flood speed in the bottom layer was 40.9 cm/sec, and the average ebb speed was 47.0 cm/sec. For comparison, the average difference between corresponding maximum flood and ebb speeds was 34.3 cm/sec in the surface layer, but only 3.5 cm/sec in the bottom layer, almost an order of magnitude smaller.

b. Lagrangian Current Measurements

Four Lagrangian current measurements were attempted using current crosses with a cross sectional area of 1.672m^2 (18 ft^2) as drogues planted in the surface and bottom layer. The purpose was to find the path of a hypothetical volume of water and its terminus at the end of the tidal cycle. Results of these 4 experiments are shown in Table 11. The first column shows

Table 11. Drogue tracking results based on launches of surface and bottom drogues at the Center Buoy, New London Dump Site. Average speed is defined as the ratio of the total distance traveled to the duration.

Approximate Tidal Stage	Date	Launch Time (EST)	NL*	Duration	Average Speed	Terminus
Beginning Flood	5 Aug 75	0640	-3.9 hrs	5 hrs 11 hrs	Surface: 72.0 cm/sec (flood) Bottom: 69.6 cm/sec (flood) 53.9 cm/sec (ebb)	LIS
Beginning Ebb	22 Jul 75	0800	-0.6 hrs	7 hrs	Surface: 77.3 cm/sec Bottom: 64.0 cm/sec	BIS
Maximum Ebb	16 Sep 75	1018	3.8 hrs	5-1/2 hrs	Surface: 36.5 cm/sec Bottom: 23.8 cm/sec	Entr. FIS DS
Maximum Flood	8 Oct 75	0900	-2.3 hrs	6 hrs	Surface: 35.3 cm/sec Bottom: 28.1 cm/sec	Entr. NLH DS

*NL = Time of launch in hours relative to nearest time of high water at New London

LIS = Long Island Sound

BIS = Block Island Sound

FIS = Entrance to Fishers Island Sound

NLH = Entrance to New London Harbor

DS = Dump Site area

the approximate stage of the tide relative to the dump site area during which the drogues were launched; the second column gives the date of the experiments (largely confined to the warmer months because the drogues were tracked by small outboard motor boats), the time of launching relative to EST (75th Meridian Time) in the 3rd column, and the time relative to the nearest time of high water at New London in the 4th column, followed by the duration of each tracking experiment, the average speed calculated from the ratio of the total distance traveled to the duration, and the terminus or end point of the experiment. Hand-held bearing compasses were used to obtain visual bearings of nearby objects as the means of navigation control.

Plots of the drogue tracks are shown in Figures 7 through 10. The longest distances traveled and the highest average speeds were by the drogues launched during the beginning of the flood (5 August 1974) and the beginning of the ebb (22 July 1975). During the flood, the drogues accelerated rapidly, particularly near Bartlett Reef. Similarly, during the ebb, the drogues accelerated rapidly when near The Race. In either case, maximum speeds were encountered when out of the dump site area. Drogues launched during the period of maximum flow remained in the general dump site area experiencing low speeds and consequently shorter distances traveled. It is interesting to note (Fig. 9 and 10) that the surface drogues launched at maximum flood and ebb ended within 2 kilometers of the same position. The difference in total travel time between the 2 drogues was only 1/2 hour; the winds during

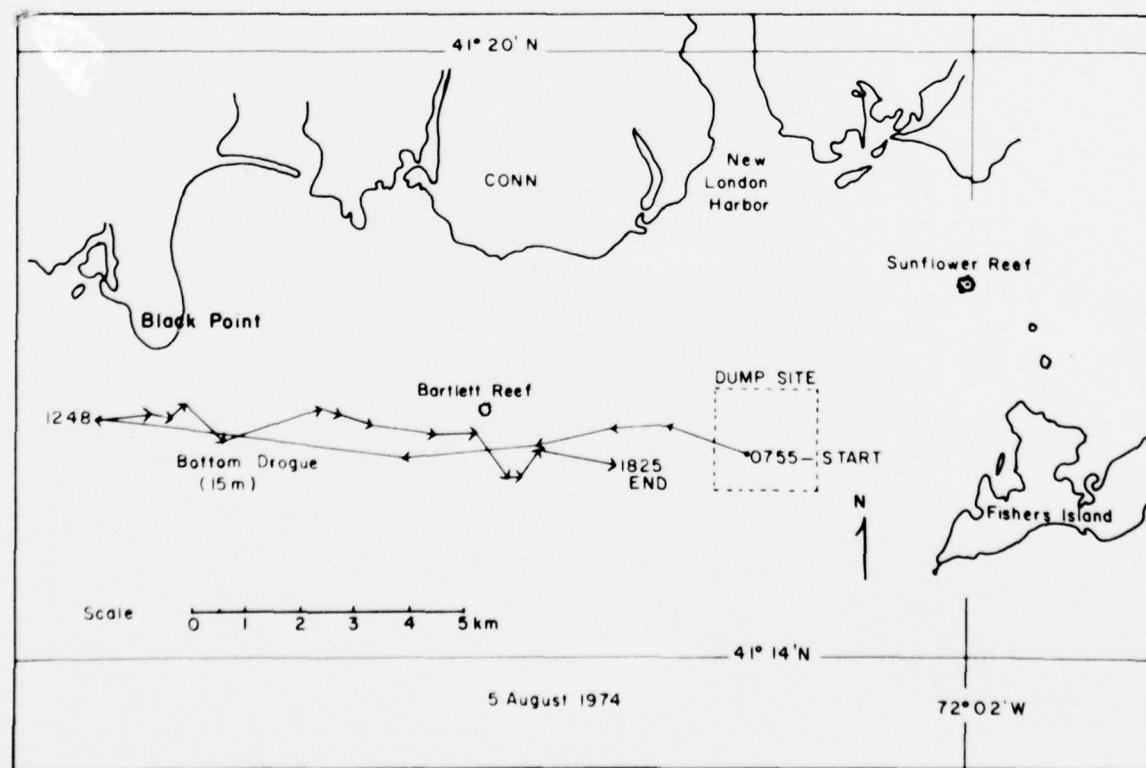
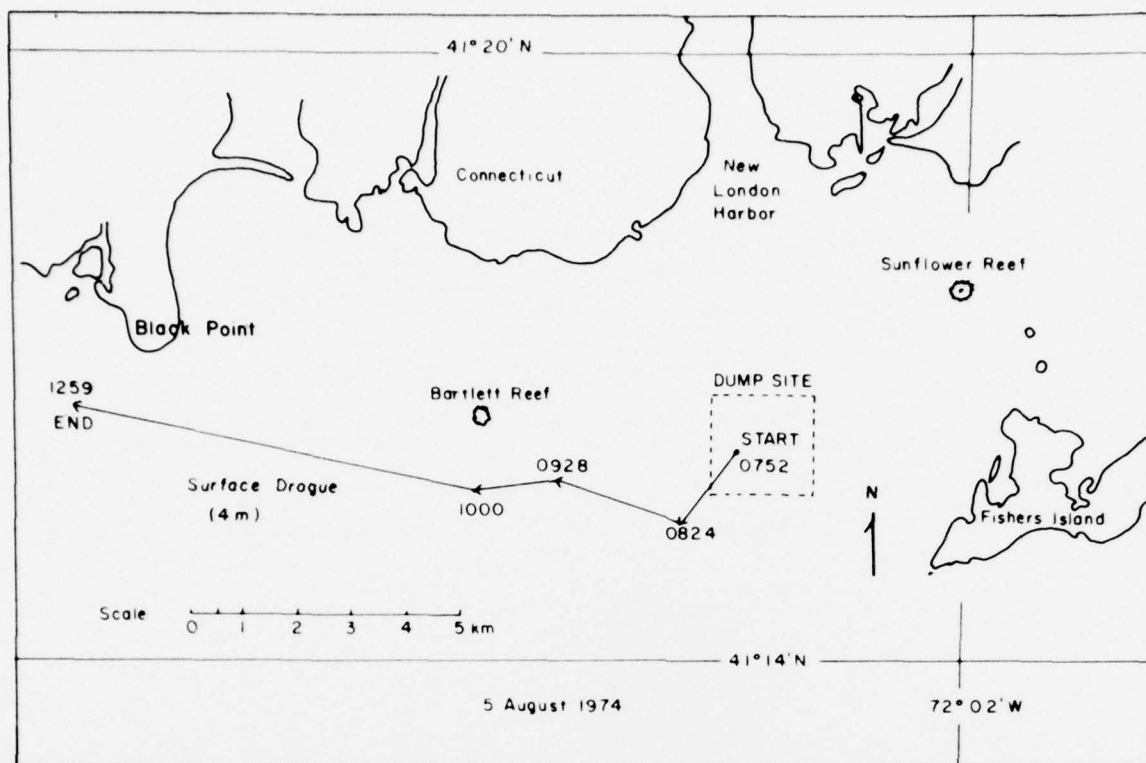


Figure 7: Displacement diagrams for surface and bottom drogues launched at the New London Dump Site on a beginning flood tide, 5 Aug. 1975.

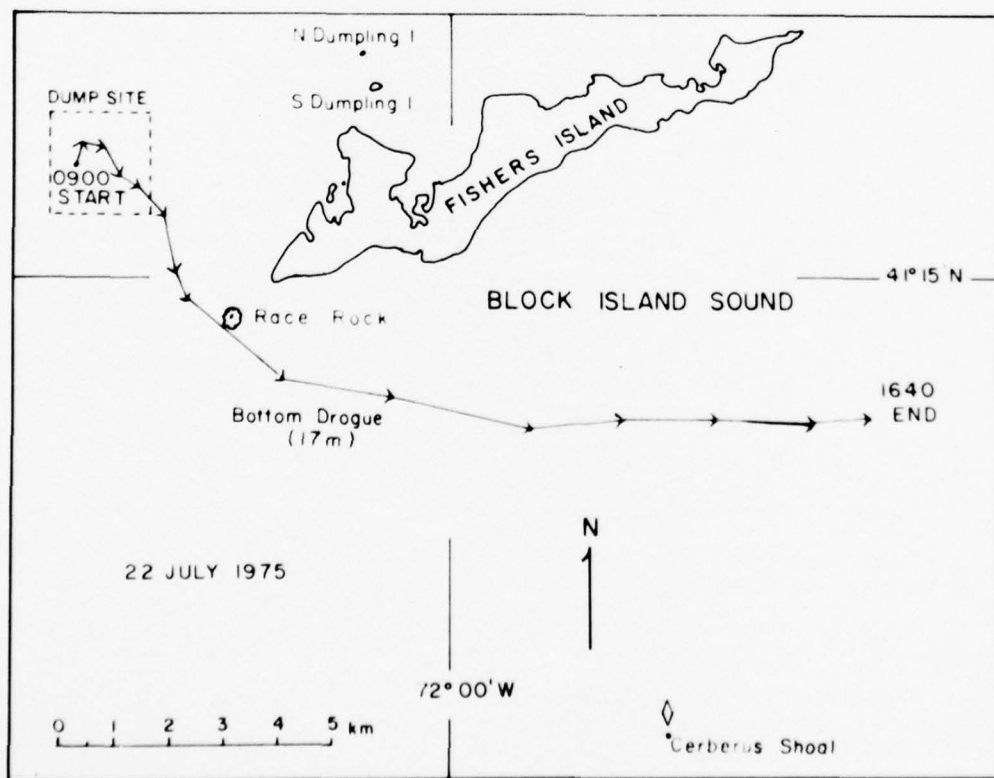
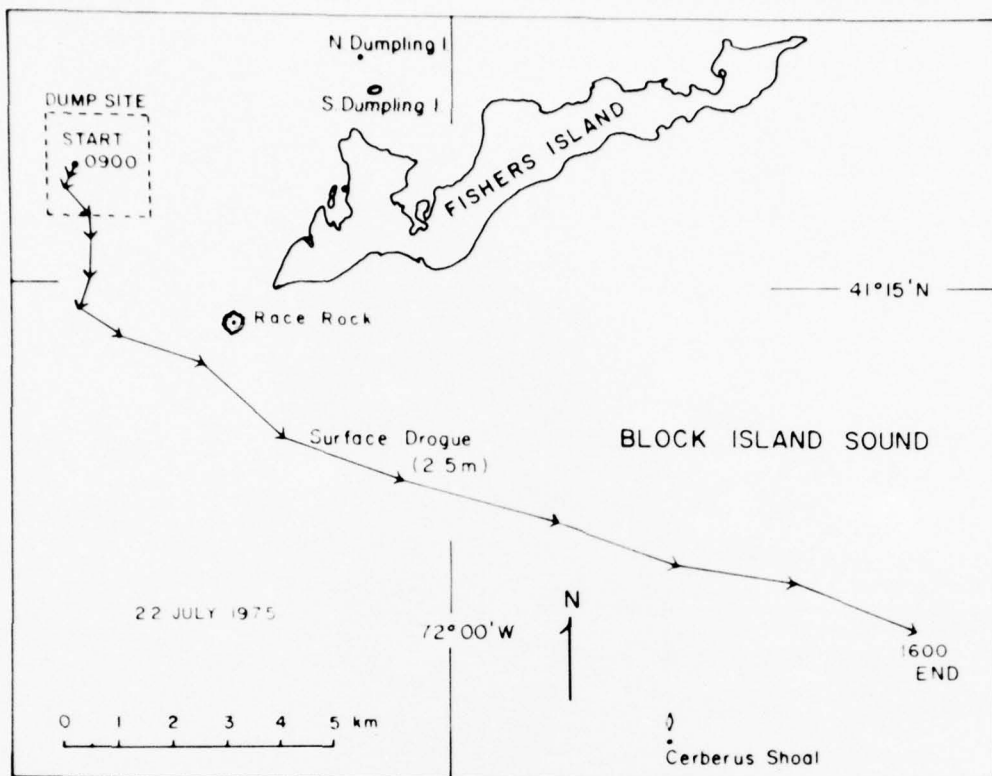


Figure 8: Displacement diagrams for surface and bottom drogues launched at the New London Dump Site on a beginning flood tide, 22 July 1975.

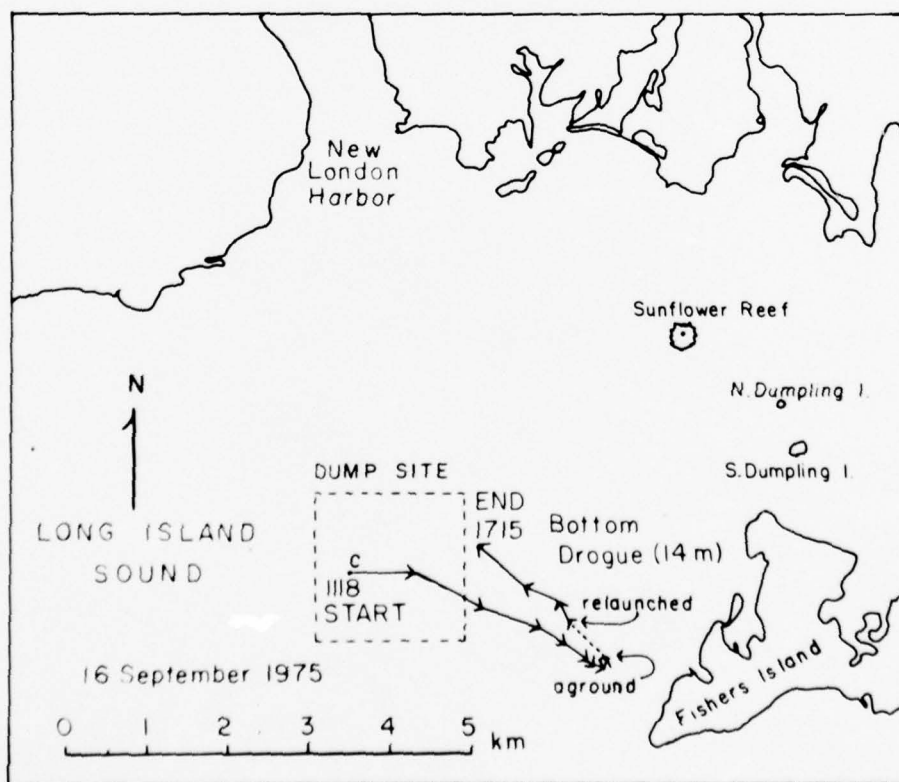
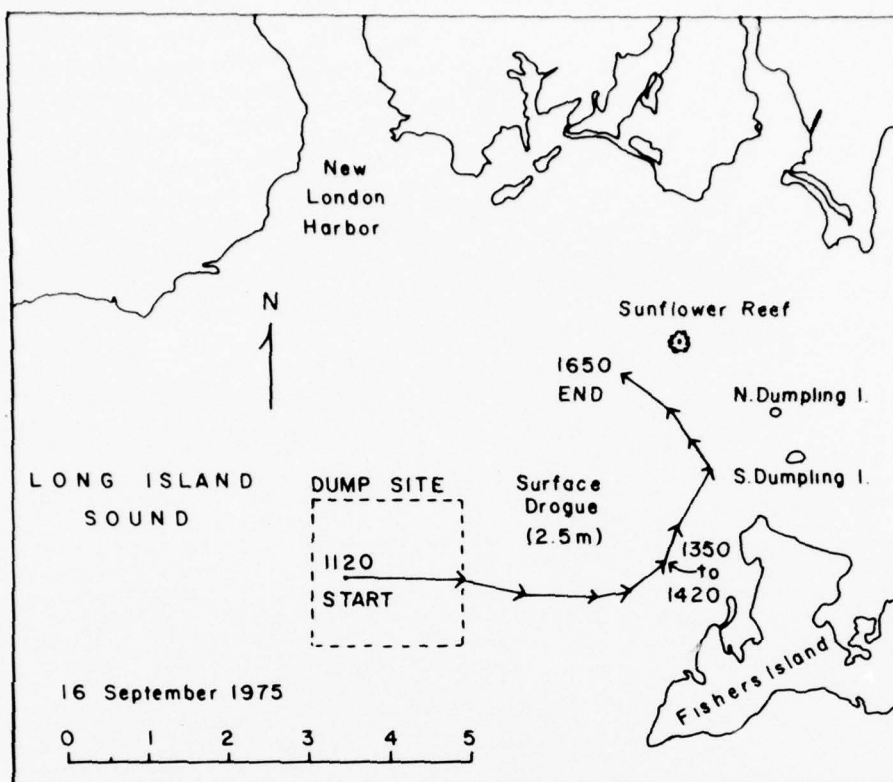


Figure 9: Displacement diagrams for surface and bottom drogues launched at the New London Dump Site close to the time of maximum ebb, 16 Sept. 1975.

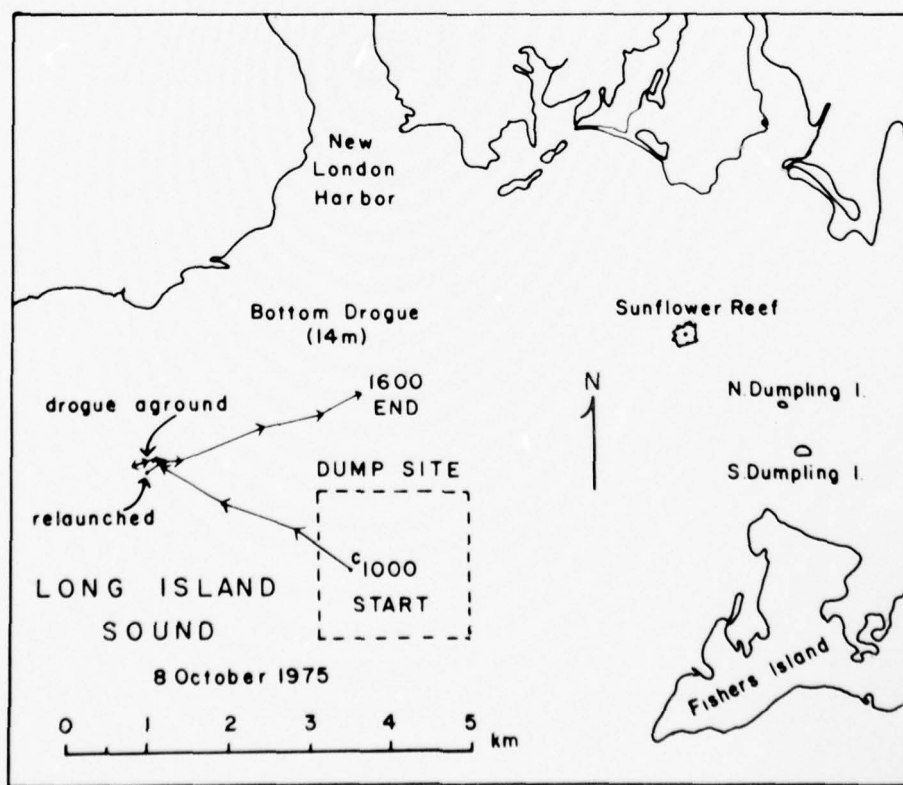
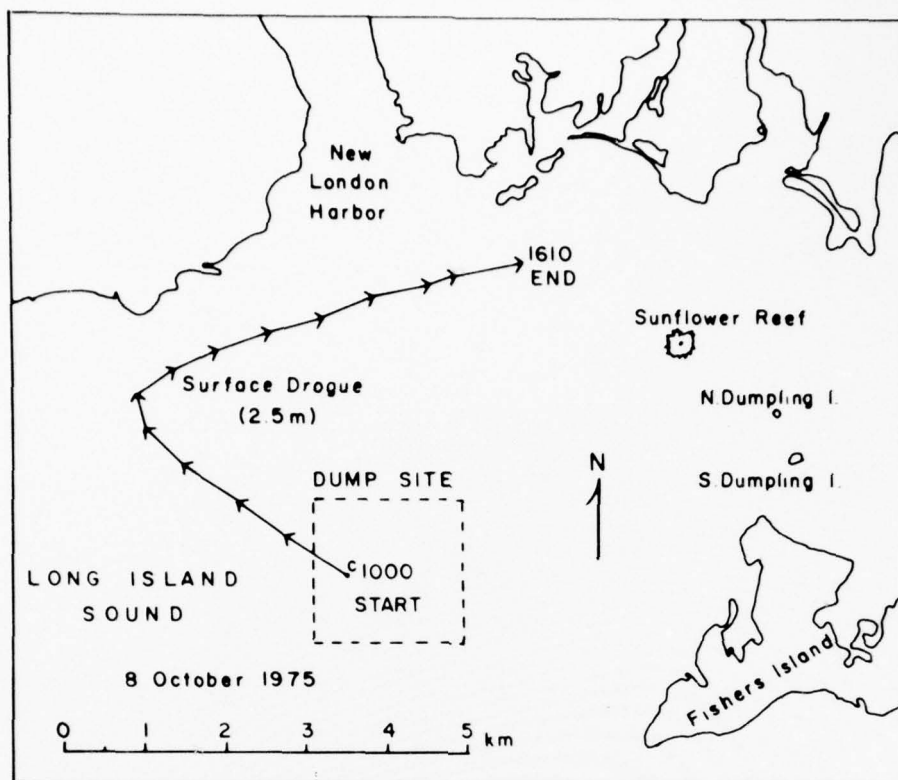


Figure 10: Displacement diagrams for surface and bottom drogues launched at the New London Dump Site close to the time of maximum flood, 8 Oct. 1975.

the Sept. 16th experiment were from the southeast at 5 to 10 mph, whereas on Oct. 8th they were west-northwest at 10 to 15 mph.

Two drogue experiments (11 Sept. 1974 and 6 Mar. 1975) were also performed in the wake of a discharging barge. Drogue depths were 2m (surface) and 15m (bottom). Results from the water quality measurements are covered elsewhere. The average speed of the surface drogue during the Sept. 11th experiment was 14.0 cm/sec; for the bottom drogue, the average speed was only 9.6 cm/sec. Both drogues remained in the dump site area; the total distance traveled by the bottom drogue was 550m and 420m by the surface drogue. In the second experiment on Mar. 6th, the average speeds were 57.3 cm/sec and 13.3 cm/sec for surface and bottom drogues respectively. Even with these higher speeds, the drogues remained in the dump site area over the sampling period.

The results from surface and bottom drifter releases at the Center Buoy are shown in Figures 11 and 12. The total number of surface drifters launched was 93 and as seen in the insert table in Figure 11, the total number of returns was 18, or 19%. Only 1 drifter was found to the west of the dump site in Long Island Sound (area NW). For the bottom drifters, a total of 89 were launched and 50 returned as of June 30, 1976, or 56% recovery. As seen in Figure 12, the major part of the returns were from the Connecticut shore to the west of the dump site area (35 drifters) as compared to 15 from the shores of Long Island. That is, of the returns,

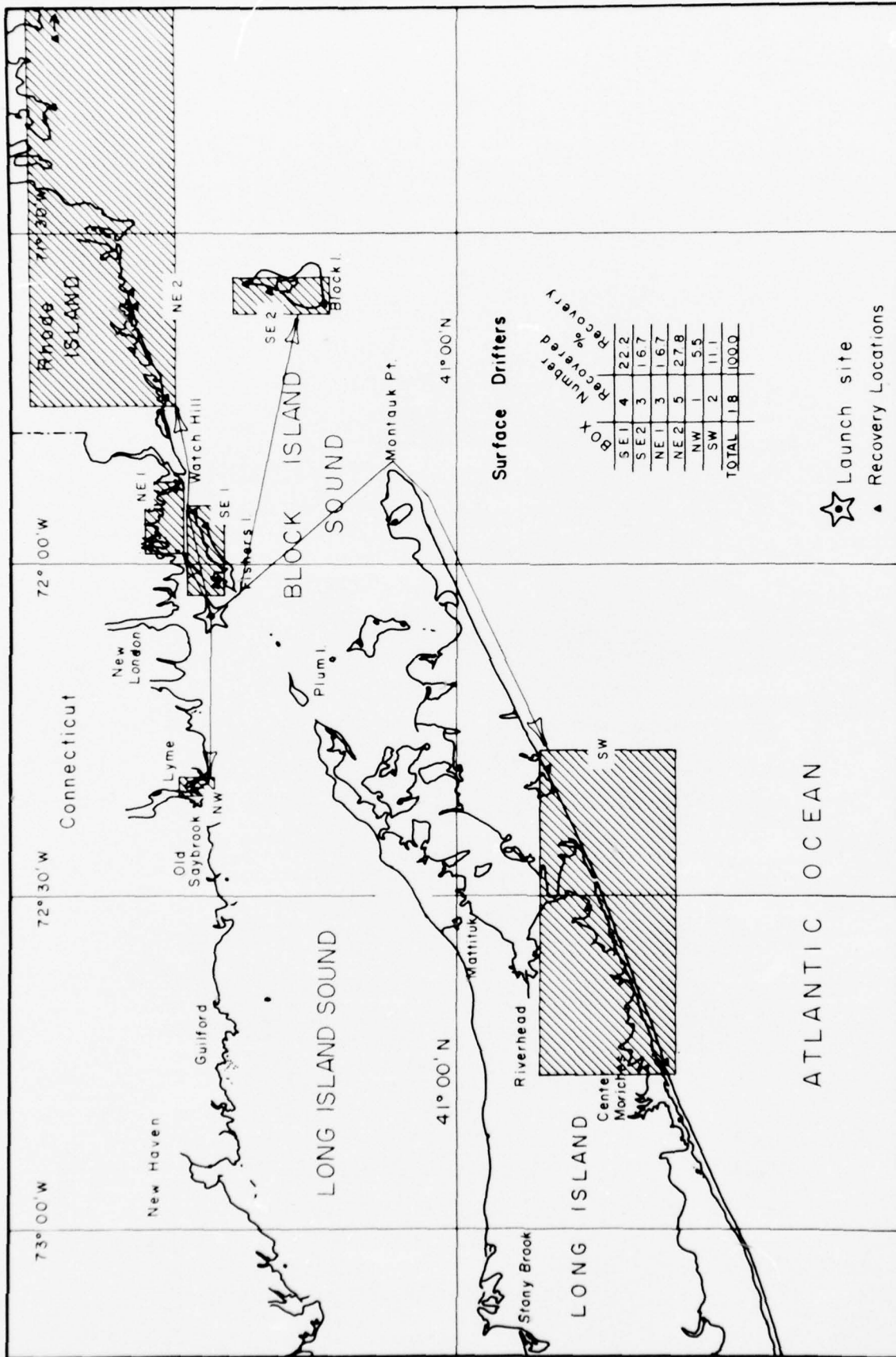


Figure 11: Location map of surface drift card returns. Insert table reflects the statistics of recovery as of June 30th, 1976. Total surface drifters launched was 93, or a 19% recovery rate.

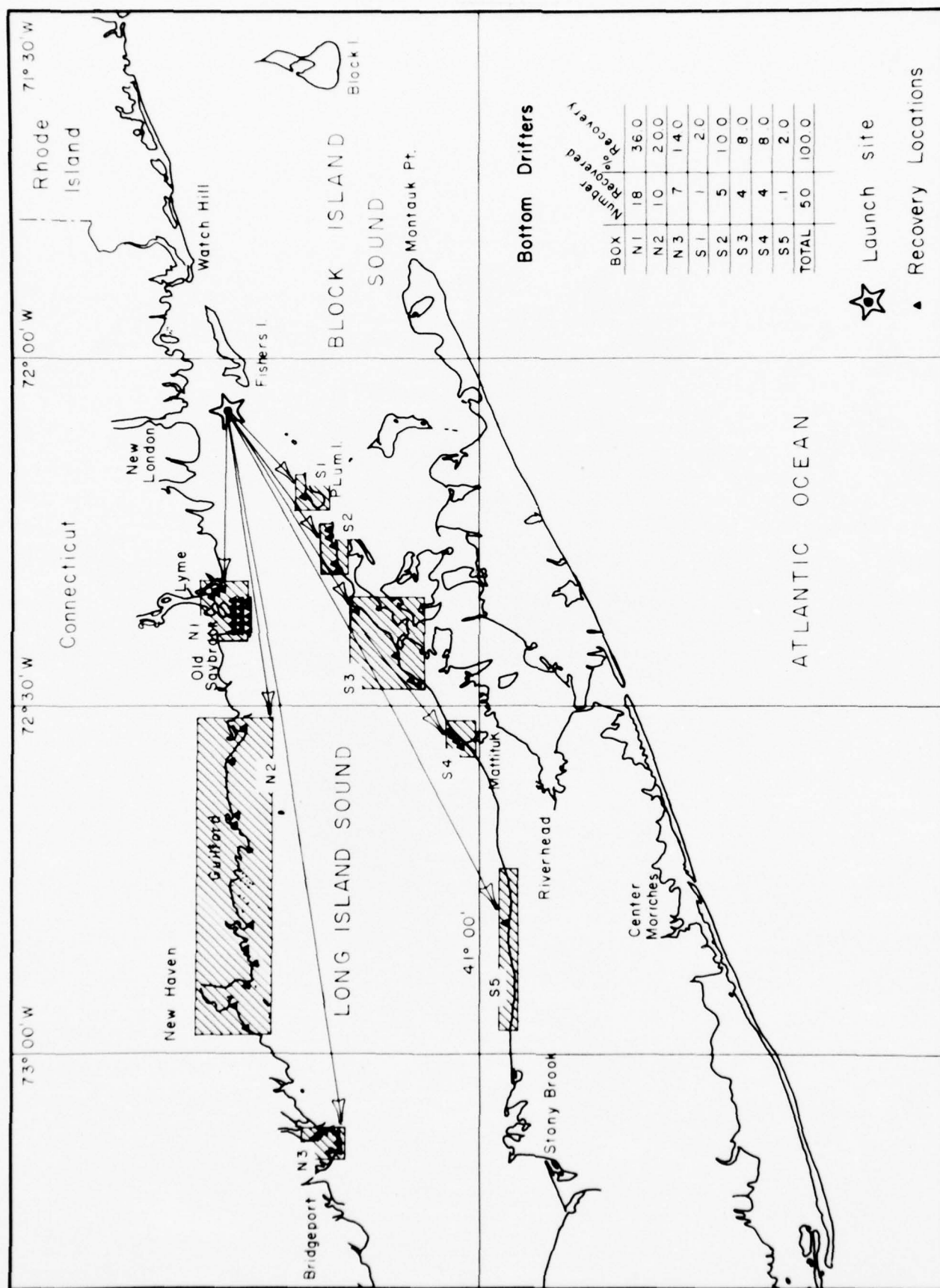


Figure 12: Location map of bottom drift card returns. Insert table reflects the statistics of recovery as of June 30th, 1976. Total bottom drifters launched was 89, or a recovery rate of 56%.

70% were from the north side of Long Island Sound (Connecticut shore) and 30% from the south side of the Sound (Long Island shore). No bottom returns were from outside of Long Island Sound.

4. Transport Calculations

Water quality parameters, particularly temperature, salinity, and suspended solids, were obtained in a quasi-synoptic fashion at the 4 corner stations, N1, E1, S1, and W1, and at the Center Buoy, CB (see Figure 1), with the purpose of estimating the transport of these properties through the dump site. The experiment was performed during an ebb tidal cycle on December 4, 1975, and repeated during the flood cycle on December 5. The sampling routine during the ebb tidal cycle was S1, CB and N1; then, E1, CB, and W1. This sequence was then repeated. On the following day during a flood tidal cycle, the sampling sequence was N1, CB, and S1; then, W1, CB, and E1. This sequence was also repeated.

The data thus obtained were then interpolated in time to correspond to concomittant current meter data at hourly (lunar) increments. Interpolations in the vertical direction were made from these temporal values that corresponded to depth for which the volume (water) transport was calculated; the product of the concentration of the water quality parameter and the volume transport yields the mass transport as defined here. The volume transport was calculated from the product of the area on a transect between isotachs per unit width. That is, isotachs were plotted for each

transect by lunar hour, and the area between isotachs determined by numerical integration; the mean velocity of the 2 isotachs was used in the product.

The results of these computations are tabulated in Table 12a. Negative values on the East/West transect define a transport to the west while negative values on the North/South transect refer to transport to the south. A zero means the value was below ± 0.5 . Highest transport values occur during the ebb tidal cycle where highest velocities and largest durations are obtained. Transport is to the south at the beginning of the ebb cycle, is north during lunar hour 4 (1335) and changes to the south again during the last stage of the ebb cycle at 1457 (hour 5).

A similar experiment was performed on 10 February 1976. The results from this cruise are tabulated in Table 12b. This experiment was terminated before the last station was made due to the onset of extreme weather conditions. Values are slightly lower in comparison to the December data and reflect in part the lower salinities and temperatures for that time of the year (winter).

DISCUSSION

The long-term effect of disposing of dredge spoil material was not expected to significantly alter existing temperature distributions either spatially or temporally, but may affect short-term salinity distributions in the immediate area of the dumping due to the definition of "salinity." Nevertheless, one would not look to salinity, or temperature as indicators of possible changes in a disposal area.

Table 12. Calculated transport of mass (water), salt, heat, and suspended solids across the East/West and North/South transects of the New London Dump Site.

a. 4 & 5 December, 1975

		East/West Transect				North/South Transect			
Tidal Stage	Time	T _v (10 ³)	T _s (10 ⁶)	T _T (10 ³)	T _{ss} (10 ⁶)	T _v (10 ³)	T _s (10 ⁶)	T _T (10 ³)	T _{ss} (10 ⁶)
Ebb - 4 Dec.	1151	14	428	154	60	-	-	-	-
	1253	19	557	200	76	-20	-42	-23	-12
	1355	20	602	214	83	-0-	15	5	-0-
	1454	17	518	186	75	-1	-17	-6	-3
Flood - 5 Dec.	0727	-14	-411	-147	-62	6	164	58	24
	0829	-13	-386	-138	-53	11	314	112	40
	0931	-0-	-10	-4	-1	10	289	103	38

b. Ebb tidal cycle, 10 February 1976

Time	East/West Transect				North/South Transect			
	T _v (10 ³)	T _s (10 ⁶)	T _T (10 ³)	T _{ss} (10 ⁶)	T _v (10 ³)	T _s (10 ⁶)	T _T (10 ³)	T _{ss} (10 ⁶)
0752	14	390	27	39	-1	-42	-3	-4
0854	14	420	28	50	-2	-66	-4	-7
0957	12	336	23	39	-3	-100	-7	-12

T_v: Volume (mass Transport x 10³) in m³/sec

T_s: Salt Transport x 10⁶ in gm/sec

T_T: Heat Transport x 10³ in cal/sec

T_{ss}: Suspended Solids Transport in mg/sec

Temperature measurements made through the year show no unusual deviation from historical data acquired at Plum Island a few miles south. In fact, the annual average temperature for the surface waters at the dump site found by averaging the values in the last column of Table 2a was 12.0°C, in agreement with the annual average reported by Garvine (1974) for eastern Long Island Sound. The corresponding average annual surface salinity was 29‰ as compared to the value of 28‰ reported by Garvine. The difference can be attributed to the proximity of the New London Dump Site to The Race and the higher salinity waters of Block Island Sound.

Of a more practical concern, however, is the resulting stability of the water column which is a function of temperature and salinity. A very simple estimate of the vertical stability, E , for shallow waters can be calculated from the vertical gradient of σ_t (Sverdrup, Johnson, and Fleming, 1942, pg. 417), i.e.,

$$E(10^{-8}) = (\Delta\sigma_t/\Delta Z) \times 10^{-5}$$

where $\Delta\sigma_t$ is the difference between the surface and bottom values of σ_t and ΔZ is the depth in meters and taken as 17m for an average. Positive values of E signify a stable water column with less dense water overlying denser water whereas negative values indicate an unstable density stratification. Table 13 shows the results of applying the approximation above to the data in Tables 2a and b. As seen in the table, all values are positive, indicating stability, albeit weak, in comparison to very stable values in the New York Bight (Hollman & Meguire, 1973) where an average

value for September was 200 for example. In the bight area, however, the stability is probably more temperature dependent than in Long Island Sound.

Table 13. Estimated stability criteria, E, for the water column calculated from the data in Tables 2a and 2b and using an average depth of 17m.

Month	E x 10 ⁸
Feb	78
Mar	46
May	79
Jul	31
Aug	43
Sep	26
Dec	65

Observed transmissivity in coastal waters is significantly influenced by terrigenous material resulting from land drainage and biological processes such as plankton blooms. Therefore, a high degree of natural variability is to be expected in the transmissivity of coastal or estuarine waters as compared to oceanic waters. Since transmissivity is path length dependent, results and discussions will center around the calculated beam attenuation coefficient, β , with the units per meter (see Definitions).

The monthly average beam attenuation coefficients presented in Table 3 are a fairly good representation of "background" values for the New London Dump Site area, noting however, that large deviations can occur

due to natural causes such as unusual plankton blooms or heavy runoff from the land, all of which depend upon external influences. Values are high in late summer in agreement with an annual distribution for extinction coefficients reported by Duxbury (1971, Table 8.2). Another secondary maximum would be expected in late spring.

In general, the beam attenuation coefficient, β , increases with depth as can be noted in Table 3. This increase with depth can be attributed to turbulence effected by the bottom tidal currents and leading to a resuspension of the bottom material. This causes a decrease in bottom transmittance and thereby increases the value of β (Jerlov, 1963 and also Gordon, 1974).

Relative spatial variability is of the order of 20% (maximum observed was 26%) as determined from the horizontal transmissometer tows (Table 4); here the estimate was calculated from the ratio of twice the standard deviation divided by the mean, or twice the coefficient of variation.

Relative temporal variations were of the order of 10% or less, calculated in a similar fashion from the data of July 17, 1975. Gordon (1974) also reported that the change in turbidity of the water column is small over a tidal cycle.

Gordon (1974) made a study of the dispersion of dredge spoil material in central Long Island Sound. His observations show that there is an outward spreading density cloud produced by the impact of the discharged material with the bottom and the figures shown by Gordon are similar to

Figure 4 of this report. The variations with time indicate that there may be more than one outward spreading density cloud produced although not of equal turbidity. Gordon found that the initial horizontal spreading of this density cloud or bottom surge, was about 12m/min.

A virtual velocity vector \vec{R}_C , can be defined for this density cloud such that

$$\vec{R}_C = \vec{R}_A - \vec{R}_B$$

where \vec{R}_A is the velocity of this cloud derived from the observed time required for the cloud to reach the transmissometer, and, \vec{R}_B is the mean bottom current velocity over the time period calculated from current meter data. The distance and direction of the discharging barge relative to the fixed transmissometer were known. In effect, \vec{R}_A is the resultant velocity between the mean bottom current, \vec{R}_B , and the virtual velocity, \vec{R}_C , of the cloud itself. Values for the speed component of \vec{R}_C are tabulated in Table 14. Directions of the virtual density cloud vectors have been omitted since for a radially outward moving cloud, the direction depends upon the relative position of the discharging barge and the transmissometer.

The average of all 8 speeds listed in Table 14 is 16.4m/min and includes of necessity, the downward flow as well as the outward flow. Variations in speeds, particularly for the 2 downstream cases may in part be due to the type of material being discharged, i.e., whether rocks, sand,

or fine clays, and to differences in the bottom currents at the time. For example, the mean bottom speed for the May 20th experiment was approximately 16m/min and 22m/min during the July 9th experiment, an increase in speed of approximately 38% while the corresponding increase in speed of the virtual density cloud velocity was 31%.

Table 14. The speed of the virtual velocities, \vec{R}_C , of the various density clouds shown in Figure 4 calculated from the difference between the observed velocity of the cloud and the average bottom currents over the time period. The average of all 8 listed speeds is 16.4m/min.

Date	Cloud No.	Approx. Time	Speed (m/min)
20 May 1975 Downstream Monitoring	1	1140	13.0
	2	1151	13.8
21 May 1975 Upstream Monitoring	1	1210	18.4
	2	1212	17.5
	3	1220	15.5
9 July 1975 Downstream Monitoring	1	1253	16.4
	2	1256	17.6
	3	1300	18.8

Linear correlations were calculated between the beam attenuation coefficient, β , and concomitant values for total suspended particles as determined by the New York Ocean Science Laboratory chemical oceanography group. For all the pairs of data that were close in time (and at the same

station), the correlation was 18.5%. This excludes surface values because of possible contamination from the atmosphere (dust) and the possibility of natural light leaking into the transmissometer, particularly when employing the 0.1m path length. This correlation was found to be significant at the 97.5% fractile. Correlations run on the data by individual cruises ranged in value from -22% up to 84%. Transmissivity is related to total suspended particles in the water column; thus the comparatively high significance associated with the low correlation; this problem probably relates to particle size for as pointed out by Clarke & James (1939) even careful filtering of coastal waters will not remove the colloidal material. It is probably this small diameter colloidal material that is absorbing much of the light energy particularly at short wave lengths (Jerlov, 1968). Harlett & Kuhn (1973), in their study did not attempt to calibrate their transmissometer in terms of suspended particle concentration since "differences in particle size distribution make this impossible." Therefore, further attempts to draw quantitative conclusions about suspended particle concentrations based on transmissometer observations would be only suspect and were not attempted.

A Fourier analysis was made of the hourly average Eulerian velocity components tabulated in Table 9. The results are tabulated in Table 15 where the tidal cycle period runs from -5 through +6 hours, a total of 12 lunar hours. The column labeled C_1 is the amplitude in cm/sec of the first harmonic with a 12 hour period; the column labeled ϵ_1 is the phase angle

in hours relative to the start of the cycle at -5 hours; and %V is the percent of the total variance of the data accounted for by the harmonic in question.

Table 15. Results of a Fourier analysis of composite data for current meters at the Center Buoy starting at -5 hrs through +6 hrs or the time of high tide at New London.

Depth	V	C_1	ϵ_1	%V
Surface	u	38.0	10.302	95
	v	9.2	2.892	42
Mid-depth	u	43.5	9.806	95
	v	15.9	2.905	62
Bottom	u	33.8	9.547	96
	v	10.5	2.968	89

As is apparent from the table, the dominant velocity component as represented by the amplitude, C_1 , is the u-component (East/West). The v-component is only 24% of the magnitude of the u-component at the surface, 36% at mid-depth, and 31% at the bottom. This small increase in the relative magnitude of the v-component with depth indicates that the sub-surface tidal currents are oriented more North/South than is the surface current. This can also be noted in the net-flows shown in Figure 6, where the mid-depth and bottom flows are progressively more to the north (cyclonic rotation).

The first harmonic account for 95% or more of the total record variance

for the u-component at all 3 levels, or, in other words, the u-component is periodic with a period of 12 hours. For the v-component, however, the second harmonic with a 6 hour period dominates in the surface accounting for 52% of the variance as compared to 42% for the first harmonic (Table 15). Therefore, there are two dominant maxima in the v-component, i.e., a northerly flow must occur twice in the tidal cycle, hence, must occur during the ebb as well as during the flood. This is also readily observed in the current meter plots (Appendix) where a shift in v from south to north occurs approximately mid-way through the ebb cycle. In the hourly averages, this occurs during hour 4 for the surface and mid-depth and during hour 5 for the bottom and can be seen as a small increase of 0.076 hours in the phase angle (ϵ) with depth.

An increase of 0.755 hours (45 minutes) with depth also occurs in the phase angle for the u-component. Thus, the maximum value for u (peak current) occurs 5.302 hours following the time of high water at New London at the surface, 4.806 hours at mid-depth, and 4.547 hours at the bottom. That is, events such as maximum flow and slack, occur earlier at the bottom than at the surface.

The average currents can readily be modelled for each of the 12 hours from the following equations where $t=1$ corresponds to -5 hours, $t= 2$ to -4 hrs,..., $t = 6$ corresponds to H, etc.,

Surface	$u(t) = 17.7 + 38.0 \cos \pi/6 (t - 10.302)$
(4m)	$v(t) = -5.6 + 9.2 \cos \pi/6 (t - 2.892) + 10.2 \cos \pi/3 (t - 4.304)$

Mid-depth (10m)	$u(t) = 17.4 + 43.5 \cos \pi/6 (t-9.806)$ $v(t) = -0.3 + 15.9 \cos \pi/6 (t-2.905) + 12.0 \cos \pi/6 (t-4.477)$
Bottom (17m)	$u(t) = 7.8 + 33.8 \cos \pi/6 (t - 0.547)$ $v(t) = 2.1 + 10.5 \cos \pi/6 (t-2.968) + 3.2 \cos \pi/3 (t-4.896).$

These rectangular components can be readily converted to polar coordinates through the definitions, viz"

$$R(t) = [u(t)^2 + v(t)^2]^{1/2}$$

and

$$\theta = \text{arc tangent } v(t)/u(t)$$

where R is the speed (cm/sec) of the current in the direction, θ . Thus, the average currents barring any unusual weather phenomenon, can be readily predicted from the above equations knowing the time of high water. This time corresponds to setting $t = 6$ in the above equations; any other time is easily found by simply adding or subtracting the appropriate hours. For example, if high water occurs at 1302 hrs, and the flow 5 hours prior is desired, then setting $t = 1$ and solving the above equations will yield the needed values relative to 0802 hrs on that day. Solutions to the above equations are tabulated in Table 16.

That the second harmonic, or 6 hour period, is the dominant factor for the v-component at the surface can be seen from the fact that the second amplitude is greater by nearly 10% (9.2 compared to 10.2).

Table 16. Theoretical velocities based on Fourier analysis of all current meter observations from the New London Dump Site relative to the time of high water at New London.

t	NL*	Surface		Mid-Depth		Bottom	
		u(t)	v(t)	u(t)	v(t)	u(t)	v(t)
1	-5	23.7	10.3	13.0	-2.2	-0.2	5.6
2	-4	4.1	-5.0	-8.0	3.6	-15.5	8.0
3	-3	-11.8	5.7	-22.3	15.8	-24.6	11.2
4	-2	-19.8	11.8	-25.9	23.5	-25.0	12.9
5	-1	-17.8	6.2	-17.8	17.1	-16.7	10.3
6	H	-6.2	-8.2	-0.4	-1.4	-1.8	3.2
7	1	11.7	-20.4	21.9	-19.5	15.7	-5.2
8	2	31.3	-21.5	42.9	-24.7	31.0	-10.2
9	3	47.2	-12.8	57.2	-15.9	40.1	-9.7
10	4	55.3	-3.6	60.8	-3.2	40.6	-5.0
11	5	53.2	-2.2	52.8	2.6	32.2	0.2
12	6	41.7	-7.2	35.3	0.2	17.3	3.5
Average		17.7	-3.9	17.5	-0.3	7.8	2.1

*NL: Time relative to the time of high water at New London

The Lagrangian measurements agree with the average velocity components. The flood conditions as shown in Figure 7 are close to hour -4 although the average speed for the surface was greater by 2.4 cm/sec than the bottom (Table 10). The reversal in the flow from a southeast condition at the surface to a northeast condition at hour 4 is indicated in the path that

the surface drogue followed on 16 Sept. in Figure 9. A possible explanation for this 6 hour period in the northerly component is compound tides and overtides that result when the tidal wave enters shallow water, i.e., Long Island Sound. Swanson (1971) found significant energy in the northerly components at periods of approximately 6 hours in current meter data from central Long Island Sound.

In line with this observed reversal in the north/south flow during the ebb is the observation that not only were the dominant returns of the bottom drifters from areas to the north and west of the dump site, but that most of these were launched during an ebb cycle. The drifter results, in general, agree with the previous results reported by Paskausky *et al.* (1972) and Hollman and Sandberg (1972).

The results of the transport calculations show a similar effect. Although transport values on the east/west transect are larger in absolute magnitude during the ebb when the direction is to the east than during the flood when the direction is westward (Table 12), as would be expected, the opposite appears to be the case for the north/south transect. Transport values on the north/south transect are larger in magnitude during the flood than during the ebb. The dominant direction for the north/south transport component is northward, or positive, during the flood, and of lesser magnitude southward (negative), during the ebb. Therefore, a net transport northward exists. In fact, the average northward transport of

suspended solids (T_{SS}) during the flood is almost a factor of 7 greater than the average southerly transport during the ebb. Similarly, the average northerly transport of salt (T_S) during the flood is greater by a factor of 17 compared to the average ebb transport to the south. The change in direction of the v-component of the velocity is also evident in the north/south transport as would be expected. In Table 12a, at 1355 hrs on Dec. 4, 1975, the transport is positive, i.e., northward during this part of the ebb, and reverts to the normal southerly direction the next hour. To summarize, the net transport was eastward along the east/west transect, and northward along the north/south transect. It must be emphasized that these average values are the hourly vectors, and therefore should not be expected to hold at any specific instant.

SUMMARY

Temperature and salinity data from the New London Dump Site area do not show any changes over the year of disposal operations that could be attributed to either the disposal operation or the dredge spoil material. Beam attenuation coefficients calculated from *in situ* observations of transmissivity are in general higher near the bottom than at the surface indicating that some scouring is taking place. This is observed throughout the area and is not restricted to the dump site itself and is due to the effect of turbulence generated by tidal currents; however, slight increases in the attenuation coefficient can be noted on the downstream side of spoil mounds. The average speed of the turbidity or density cloud, resulting from barge discharges was approximately 16m/min.

A reversal from a southeasterly flow to a more northeasterly flow approximately mid-way in the ebb tidal cycle is apparent from observed current meter results obtained at the Center Buoy. Average hourly velocities relative to the time of predicted high water at New London were calculated from current meter observations and used as a foundation for a predictive model. The net flow, based on an integration of observed data, was SE at the surface, E at mid-depth, and NE near the bottom. Drogue experiments tend to confirm the current meter observations. Maximum transport was in the East/West direction with highest values occurring during the ebb; for the North/South transport, higher values obtained during the flood cycle.

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D. Chemical Oceanography of Disposal Area
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INTRODUCTION

The chemical oceanography study of the New London Dump Site area was designed to detect and monitor the effects, if any, of the dumping of dredge spoils on chemical parameters in the disposal area and its environs. The study may be divided into three aspects.

One part of the study involved the water quality of the area. The parameters studied were Eh, pH, turbidity, salinity, dissolved oxygen, and suspended and volatile solids. Experiments were designed to monitor the parameters seasonally, and to observe their changes following actual spoilings.

Another aspect of the study was the analysis of the sediments at, and in the vicinity of the dump area. The parameters investigated were: chemical oxygen demand (COD), Kjeldahl nitrogen (K-N), total phosphorus, and the metals: cadmium, copper, iron, lead, nickel, zinc, and mercury. The purpose of this study was to detect any spreading of the spoil mound or of its components.

The third part of the project was designed to study the effect of dumping on benthic organisms and the seston in the area. The parameters investigated here were the same as those for the sediment study with the exception of COD.

METHODS

Water Quality Studies

Water samples for mid-depth and bottom analysis were collected using Niskin samplers. Depth of the samples was determined by using a meter wheel cable pulley with corrections made for wire angle. Surface samples were collected by means of a polyethylene bucket.

pH, Eh, and turbidity of the samples were determined in the field. The method used for pH is that described in the EPA manual (EPA, 1974). Eh was determined with a platinum electrode in accordance with the ASTM procedures for oxidation-reduction potentials of water (ASTM, 1974). Turbidity was measured by means of a Hach Turbidimeter (Model 2100A), following the manufacturer's instructions. Suspended solids were determined using approximately two liters of water, filtered on board the sampling vessel immediately upon collection. Detailed descriptions of the method may be found in Standard Methods (APHA, 1971). Dissolved oxygen was determined by the Winkler method as described in the EPA Manual (EPA, 1974). A Beckman Model #915 Total Organic Carbon Analyzer was used according to the manufacturer's directions to obtain organic content of the samples. These samples were stored frozen, until analyzed.

Seston Samples

Seston samples were collected by towing a 363 μ mesh net with a 3/4 meter opening at three depths. The collected samples were then freeze-dried, homogenized, and stored for analysis.

For a description of digestion methods, see "Marine Pollution Monitoring" (1972). Determination of heavy metals was done by flame atomic absorption spectroscopy. Mercury was analyzed by the flameless method described in Marine Pollution Monitoring, following an acid-permanganate digestion. A Perkin-Elmer Model #503 Atomic Absorption Spectrophotometer was used for all metal determinations except for some early mercury determinations which were analyzed on a Beckman Model #440 AA. Phosphorus was determined on an aliquot of the heavy metal digestion following the procedure given in Strickland (1968). Nitrogen was analyzed using an Orion Specific Ion Ammonia Probe following a Kjeldahl digestion, as described in the EPA manual (1974).

Sediment Samples

Sediment samples were collected by the Sandy Hook Marine Laboratory as described in a previous section. The samples were stored and transported to Montauk frozen. Upon receiving the samples, they were thawed, homogenized, and a portion of the wet sample refrozen and held for COD analysis. The remainder of the sample was air dried at 60°C, sieved through a 2mm non-metallic screen and held for determination of acid soluble metals, total phosphorus and Kjeldahl nitrogen.

COD determinations were performed as described in "Standard Methods" (APHA, 1971). Sediments for heavy metal analysis were acid extracted with 4M HNO_3 -0.7M HCl as described by Oliver (1973), and determined by flame

atomic absorption spectroscopy as described above. The determination of Hg was performed by flameless AA after an acid-permanganate extraction. Phosphorus was determined as described by Strickland (1968) after total digestion (Marine Pollution Monitoring, 1972). Samples were digested for the nitrogen determination by the Kjeldahl method (EPA, 1974).

Benthic Organisms

Samples of benthic organisms were collected by the Sandy Hook Marine Laboratory as described previously. These were stored frozen.

The specimens were freeze dried individually and, depending on their size, they were either combined, analyzed individually, or divided for multiple analyses after homogenization. The samples were digested and analyzed as described above for the seston samples.

Sampling Stations

Sampling stations for the water quality and seston studies are shown in Figure 1. Their coordinates are given in Table 1. The stations for sediment and benthic sampling are shown in Figure 2.

RESULTS

Areal Survey of Water Quality

Periodic surveys of the water at the dump site and its vicinity (see Figure 1 for sampling stations) were carried out for these parameters: dissolved oxygen, suspended and volatile solids, pH, Eh, turbidity, and

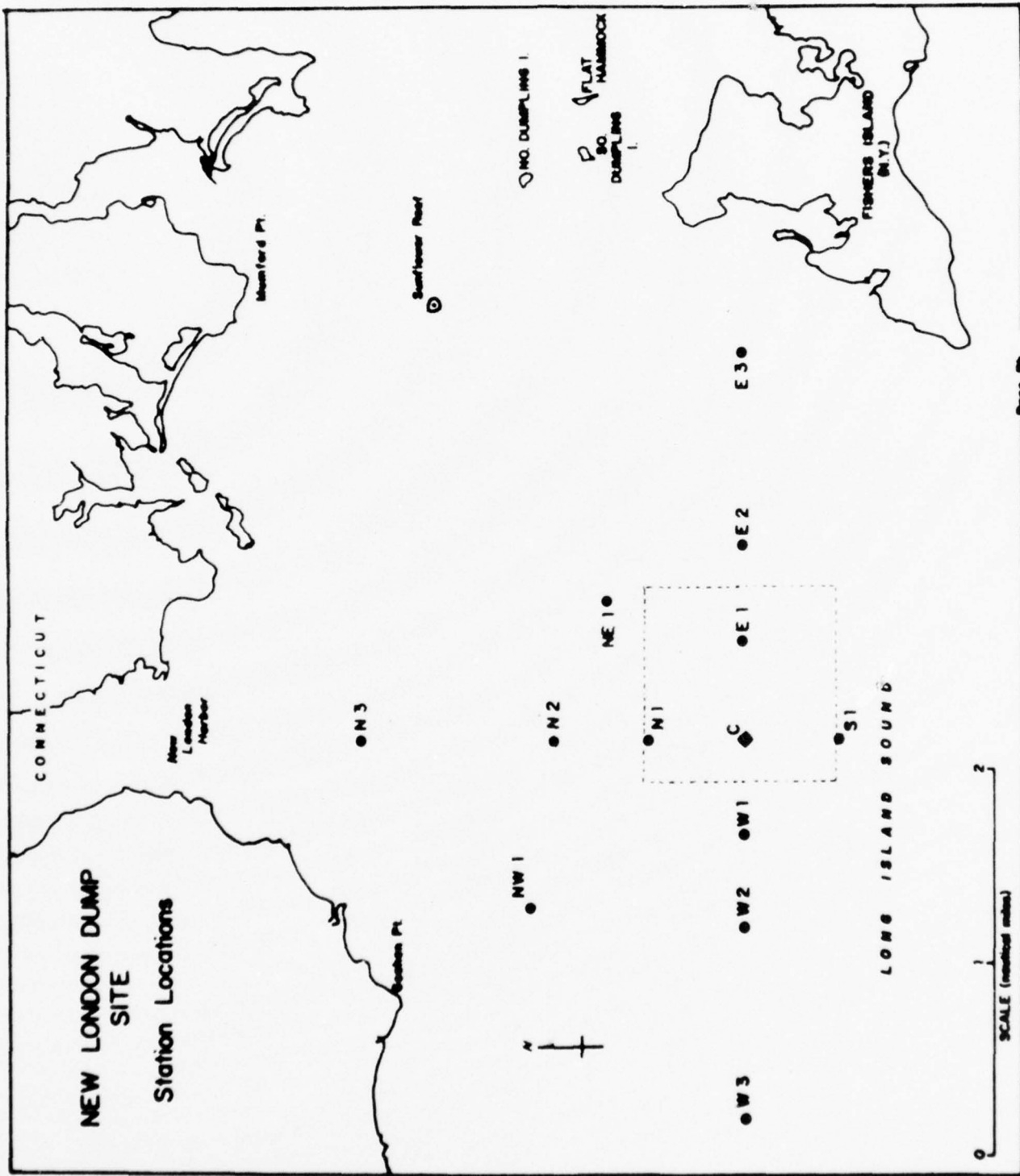
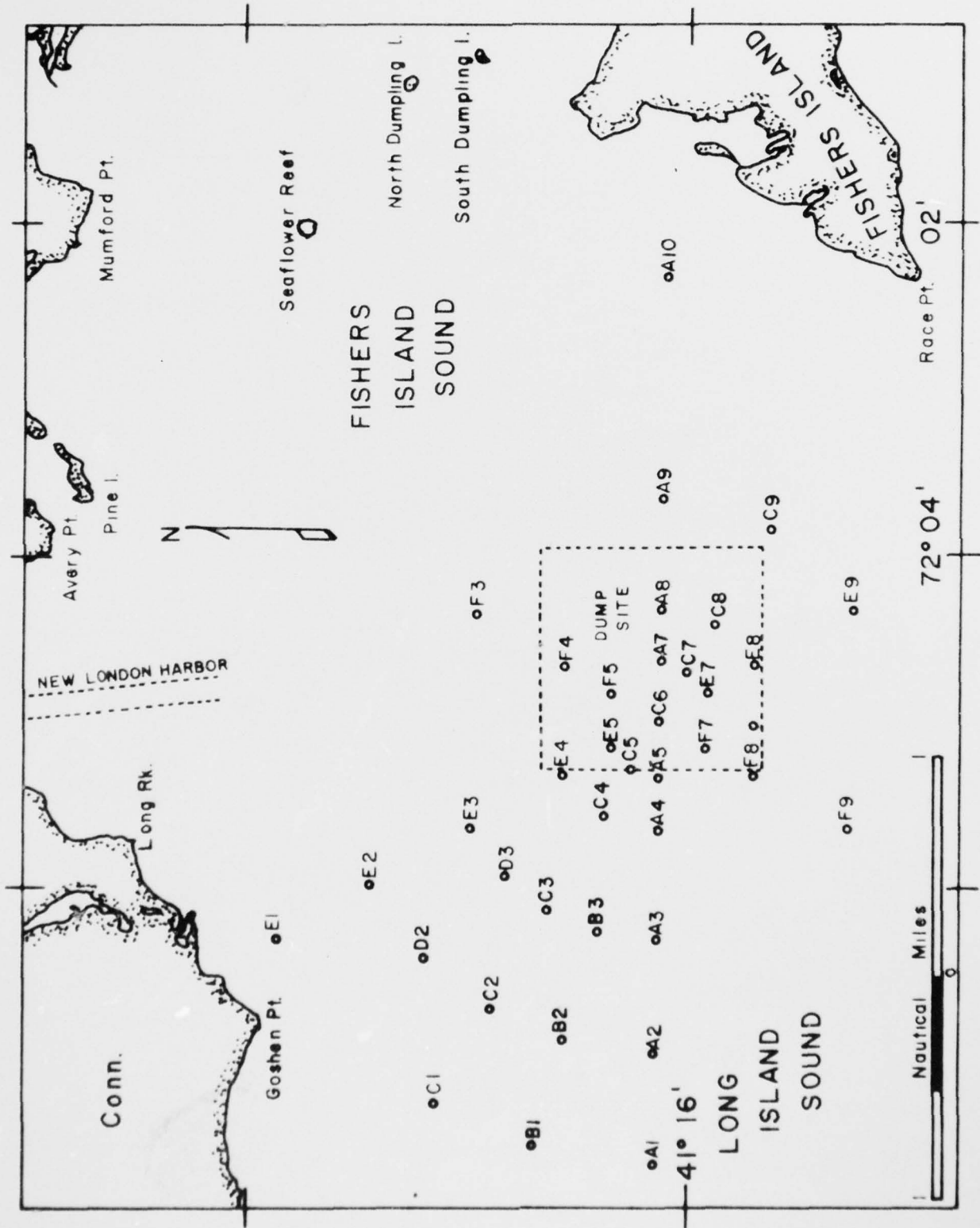


Fig. 1. Station Locations - New London Dump Site - Water Column

Table 1 : Station locations in the area of the New London Dump Site

Station	Latitude	Longitude
Center Buoy	41°16'08"	72°05'00"
W1	"	72°05'40"
W2	"	72°06'30"
W3	"	72°07'37"
E1	"	72°04'20"
E2	"	72°03'37"
E3	"	72°02'17"
S1	41°15'38"	72°05'00"
N1	41°16'38"	"
N2	41°17'08"	"
N3	41°18'08"	"
NW1	41°17'14"	72°06'08"
NE1	41°16'50"	72°04'04"

Figure 2: Sediment station locations New London Dump Site



organic carbon. These areal surveys were designed to detect any differences in the parameters as a function of distance from the dump site. They were performed as synoptically as possible under a variety of tidal and seasonal conditions.

In general the parameters showed no trends with respect to location relative to the center station. There are variations between sampling cruises as shown in Tables 2 through 7 and Figures 3 through 8. These variations are apparently seasonal as demonstrated by a comparison with 1974 and 1975 data.

Mean dissolved oxygen values (Fig. 3) ranged from about 80% to 100% saturation. NAVOCEANO (1973) reported dissolved oxygen values ranging from 89.8 to 97.4% for 6 month averages in waters in the New London area in 1972. Seasonal changes were not noted, however.

No spatial variations appear consistently in the value of suspended solids in the dump area and adjacent waters; however, variations occurred over the entire area on a seasonal basis. Values appear to be somewhat higher during the winter months. Suspended solids are often higher in the bottom waters than in the upper waters (Table 4 and Figure 4). This observation, as well as the general magnitude of the results, are consistent with data reported by Bohlen (1975).

Turbidity values (Table 6 and Figure 5) are also generally, but not

Table 2
Average Values for Volatile Solids (percent total solids)

	31 July 1974	11 Sept 1974	11 Dec 1974	28 Feb 1975	21 May 1975	30 July 1975*	26 Aug 1975	23 Oct 1975	4 Dec 1975	5 Dec 1975	10 Feb 1976	10 June 1976	5 Aug 1976
S	79.92 ±6.38	48.57 ±5.56	28.20 ±12.41	17.62 ±5.62	63.36 ±18.80	49.91 ±7.94	25.30 ±9.25	16.55 ±3.74	16.40 ±2.30	19.87 ±2.58	28.54 ±7.43	31.58 ±16.68	27.82 ±7.27
M	70.23 ±10.63	44.00 ±9.53	18.80 ±7.49	11.62 ±4.56	58.50 ±23.20	21.41 ±7.52	14.38 ±6.27	N.D.	12.40 ±1.94	14.25 ±1.98	23.90 ±8.60	31.99 ±5.39	18.51 ±4.70
B	61.28 ±9.16	30.14 ±5.42	14.90 ±3.78	10.50 ±3.81	54.90 ±22.24	13.91 ±1.97	13.23 ±5.58	13.87 ±2.16	10.40 ±1.14	11.37 ±1.59	19.45 ±3.58	28.36 ±4.87	16.20 ±5.72

Table 3
Average Values of Dissolved Oxygen (percent saturation)**

	31 July 1974	11 Sept 1974	11 Dec 1974	28 Feb 1975	21 May 1975	30 July 1975*	26 Aug 1975	4 Dec 1975	5 Dec 1975	10 Feb 1976	10 June 1976	5 Aug 1976
S	100.15 ±2.16	83.65 ±4.14	91.44 ±1.76	95.50 ±1.21	96.65 ±0.98	109.18 ±3.41	93.00 ±3.97	90.72 ±4.45	92.24 ±1.62	N.S.	106.85 ±6.27	93.19 ±11.78
M	95.63 ±1.50	80.40 ±5.22	91.44 ±0.94	93.95 ±1.02	97.20 ±2.00	91.13 ±1.98	90.18 ±0.82	90.72 ±3.12	92.84 ±1.42		103.60 ±6.30	89.74 ±9.26
B	93.68 ±1.14	80.40 ±5.36	91.68 ±1.51	93.47 ±1.14	96.48 ±2.05	90.04 ±1.75	85.10 ±17.84	90.72 ±3.84	92.24 ±1.19		102.58 ±4.44	86.17 ±12.69

*Alternate Dump Site

**Upon re-examination of previous analyzed data, an error in calculation was found. This resulted in DO's which are 20% higher than originally reported. This affects all DO's reported before the January through March 1976 report.

Table 4
Average Value of Suspended Solids

	31 July 1974	11 Sept 1974	11 Dec 1974	28 Feb 1975	6 March 1975	21 May 1975	30 July 1975*	26 Aug 1975	23 Oct 1975	4 Dec 1975	5 Dec 1975	10 Feb 1976	10 June 1976	5 Aug 1976
S	2.20 ±0.43	1.77 ±0.28	3.39 ±0.41	2.54 ±0.55	2.52 ±0.30	1.35 ±0.32	1.05 ±0.16	2.08 ±0.56	2.93 ±0.68	3.12 ±0.36	3.18 ±0.47	2.59 ±0.31	2.62 ±0.57	2.67 ±0.65
M	2.17 ±0.47	2.17 ±0.37	4.11 ±1.61	4.41 ±0.76	3.32 ±1.95	1.13 ±0.15	1.94 ±0.37	2.08 ±0.57	N.D.	4.15 ±0.48	3.61 ±0.74	3.12 ±0.40	2.74 ±0.53	2.30 ±0.36
B	2.75 ±0.58	3.38 ±0.71	6.54 ±2.22	5.91 ±3.72	11.00 ±13.81	1.12 ±0.18	4.19 ±1.47	2.46 ±0.71	5.06 ±2.40	5.66 ±0.70	4.68 ±0.79	3.86 ±0.37	3.21 ±0.71	2.95 ±0.69

Table 5
Average Eh Values

	31 July 1974	11 Sept 1974	11 Dec 1974	28 Feb 1975	21 May 1975	30 July 1975*	26 Aug 1975	4 Dec 1975	5 Dec 1975	10 Feb 1976	10 June 1976	5 Aug 1976
S	191.38 ±3.27	237.5 ±27.40	191.0 ±17.40	189.25 ±17.31	146.54 ±17.16	223.66 ±11.44	174.53 ±35.17	226.40 ±8.56	202.40 ±2.96	147.00 ±6.40	199.66 ±49.71	N.D.
M	191.64 ±3.12	235.5 ±10.26	196.23 ±19.67	191.44 ±15.93	149.27 ±17.10	224.66 ±7.70	171.46 ±35.77	226.00 ±9.02	202.00 ±1.73	148.20 ±6.90	187.41 ±54.92	N.D.
B	191.84 ±2.88	235.12 ±14.45	193.00 ±16.97	203.85 ±16.91	149.81 ±17.90	219.58 ±14.27	169.46 ±40.38	224.80 ±8.46	203.40 ±1.51	147.60 ±7.50	197.33 ±47.88	N.D.

*Alternate Dump Site

Table 6
Average Values of Turbidity

	31 July 1974	11 Sept 1974	11 Dec 1974	28 Feb 1975	21 May 1975	30 July 1975*	26 Aug 1975	4 Dec 1975	5 Dec 1975	10 Feb 1976	10 June 1976	5 Aug 1976
S	N.D.	1.06 ±0.99	1.65 ±0.36	1.20 ±0.54	1.43 ±0.30	1.98 ±0.49	1.08 ±0.24	1.21 ±0.25	1.23 ±0.14	1.19 ±0.15	0.94 ±0.33	N.D.
M	N.D.	1.27 ±1.35	1.61 ±0.22	1.72 ±0.86	1.44 ±0.53	2.29 ±0.65	1.48 ±0.60	1.42 ±0.27	1.26 ±0.32	1.39 ±0.13	1.10 ±0.28	N.D.
B	N.D.	1.65 ±1.66	2.36 ±0.61	2.55 ±1.66	1.48 ±0.71	3.49 ±0.80	1.70 ±0.58	1.66 ±0.25	1.46 ±0.36	1.64 ±0.18	1.11 ±0.24	N.D.

Table 7
Average pH Values

	31 July 1974	11 Sept 1974	11 Dec 1974	28 Feb 1975	21 May 1975	30 July 1975*	26 Aug 1975	23 Oct 1975	4 Dec 1975	5 Dec 1975	10 Feb 1976	10 June 1976	5 Aug 1976
S	7.93 ±0.03	7.82 ±0.07	8.08 ±0.08	8.03 ±0.03	7.98 ±0.07	8.19 ±0.11	7.93 ±0.23	N.D.	8.12 ±0.04	8.19 ±0.03	8.01 ±0.15	7.67 ±0.33	N.D.
M	7.95 ±0.03	7.81 ±0.04	8.06 ±0.07	8.07 ±0.03	8.02 ±0.06	8.08 ±0.10	7.90 ±0.32	N.D.	8.14 ±0.03	8.20 ±0.02	7.95 ±0.04	7.79 ±0.25	N.D.
B	7.94 ±0.03	7.80 ±0.04	8.12 ±0.13	8.07 ±0.04	8.01 ±0.08	8.05 ±0.10	7.96 ±0.23	N.D.	8.12 ±0.04	8.17 ±0.05	7.97 ±0.21	7.83 ±0.19	N.D.

* Alternate Dump Site

Figure 3 : Average dissolved oxygen vs time
(% saturation)

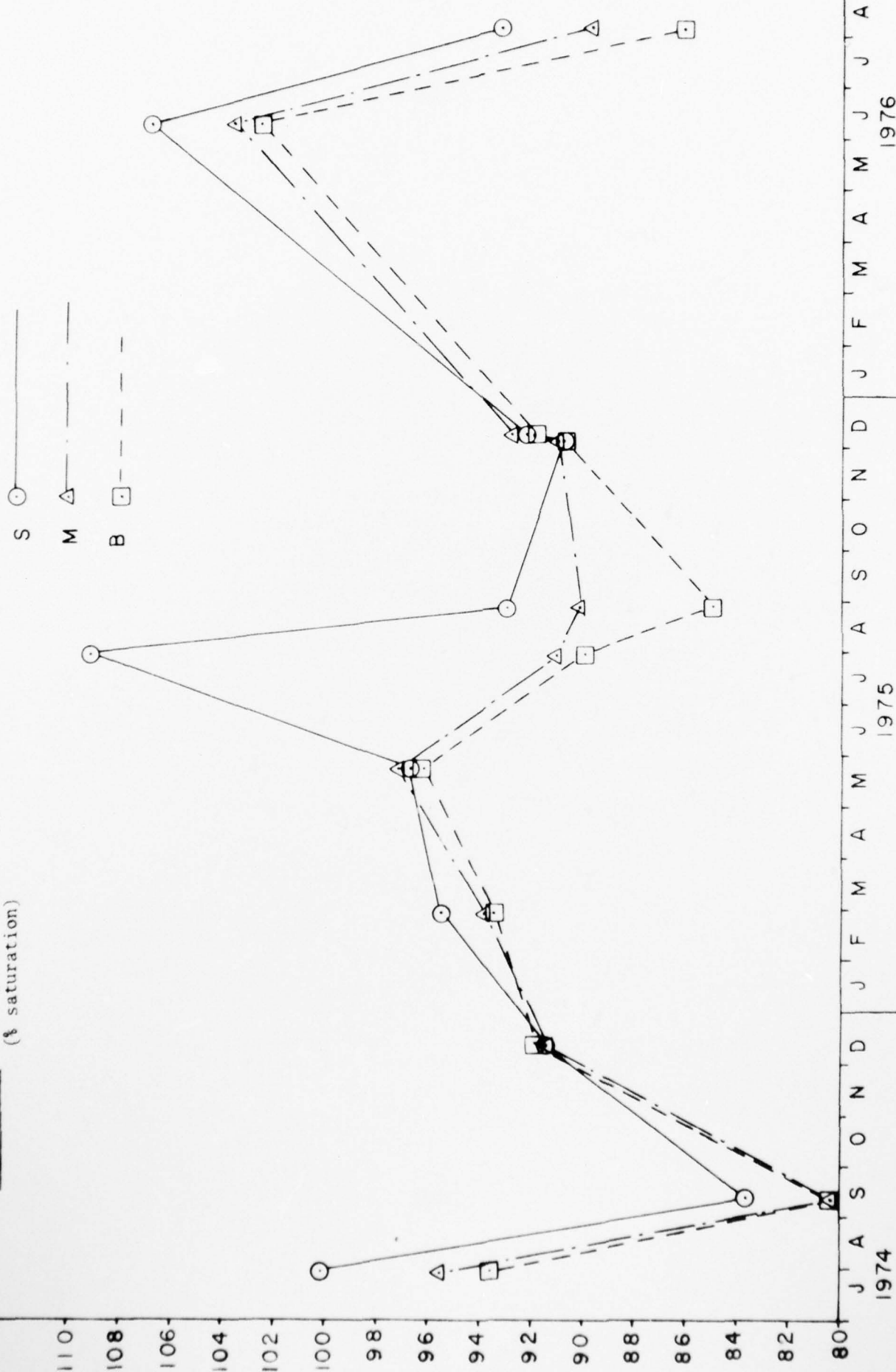


Figure 4 : Average total suspended solids vs time (mg/ℓ)

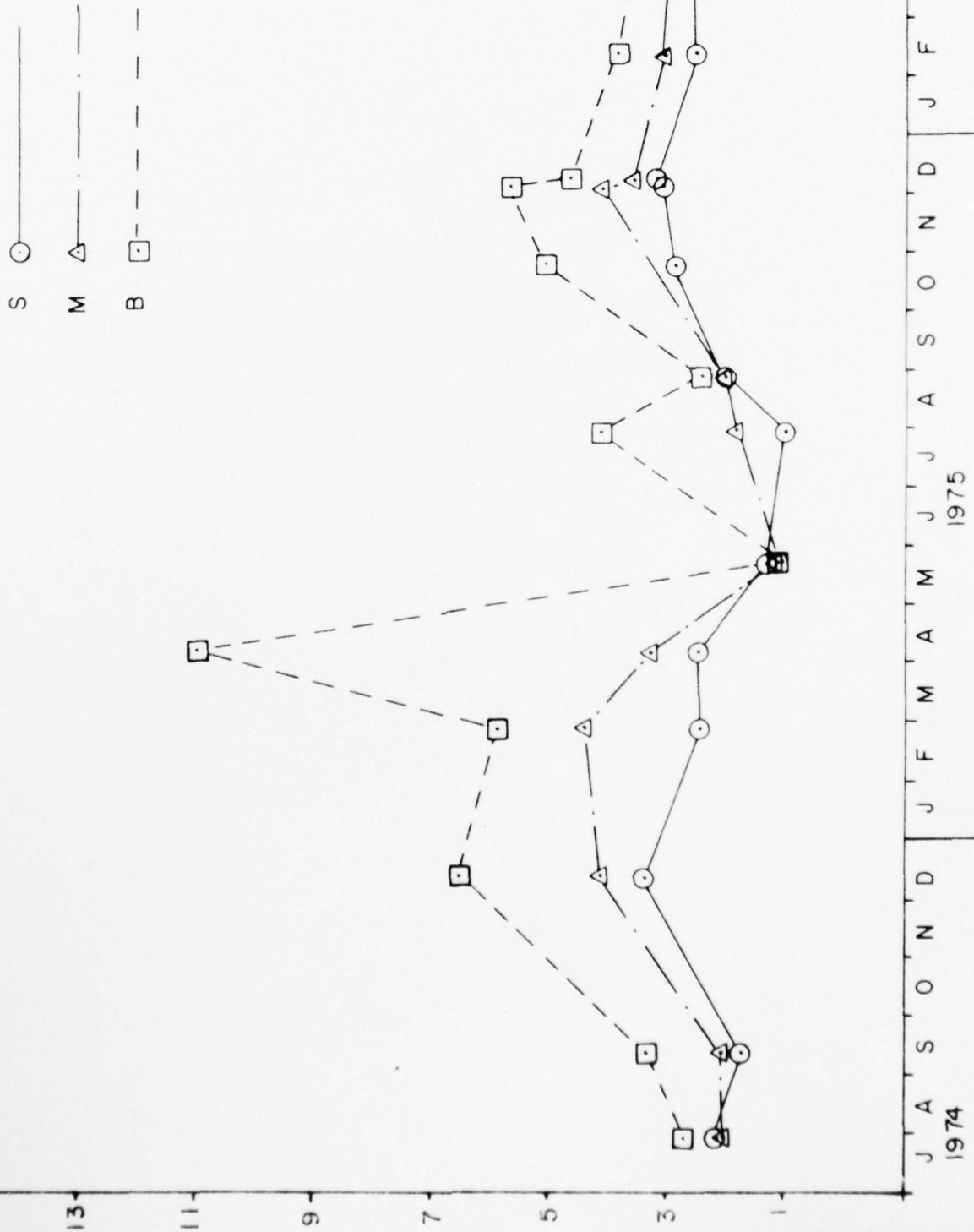


Figure 5 : Average turbidity vs time (turbidity in F.T.U.'s)

S ○ —
 M △ —
 B □ - -

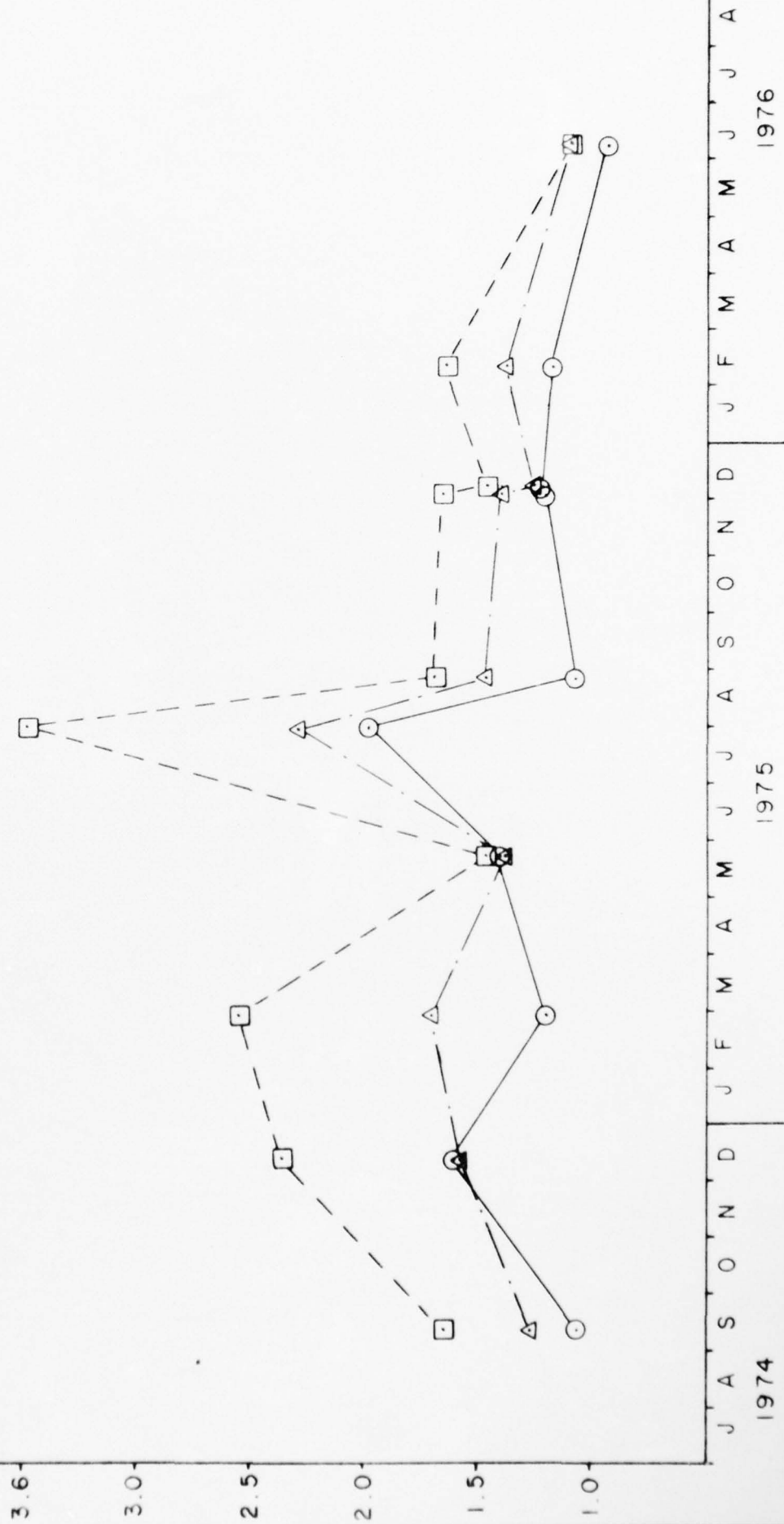


Figure 6 : Average pH vs time

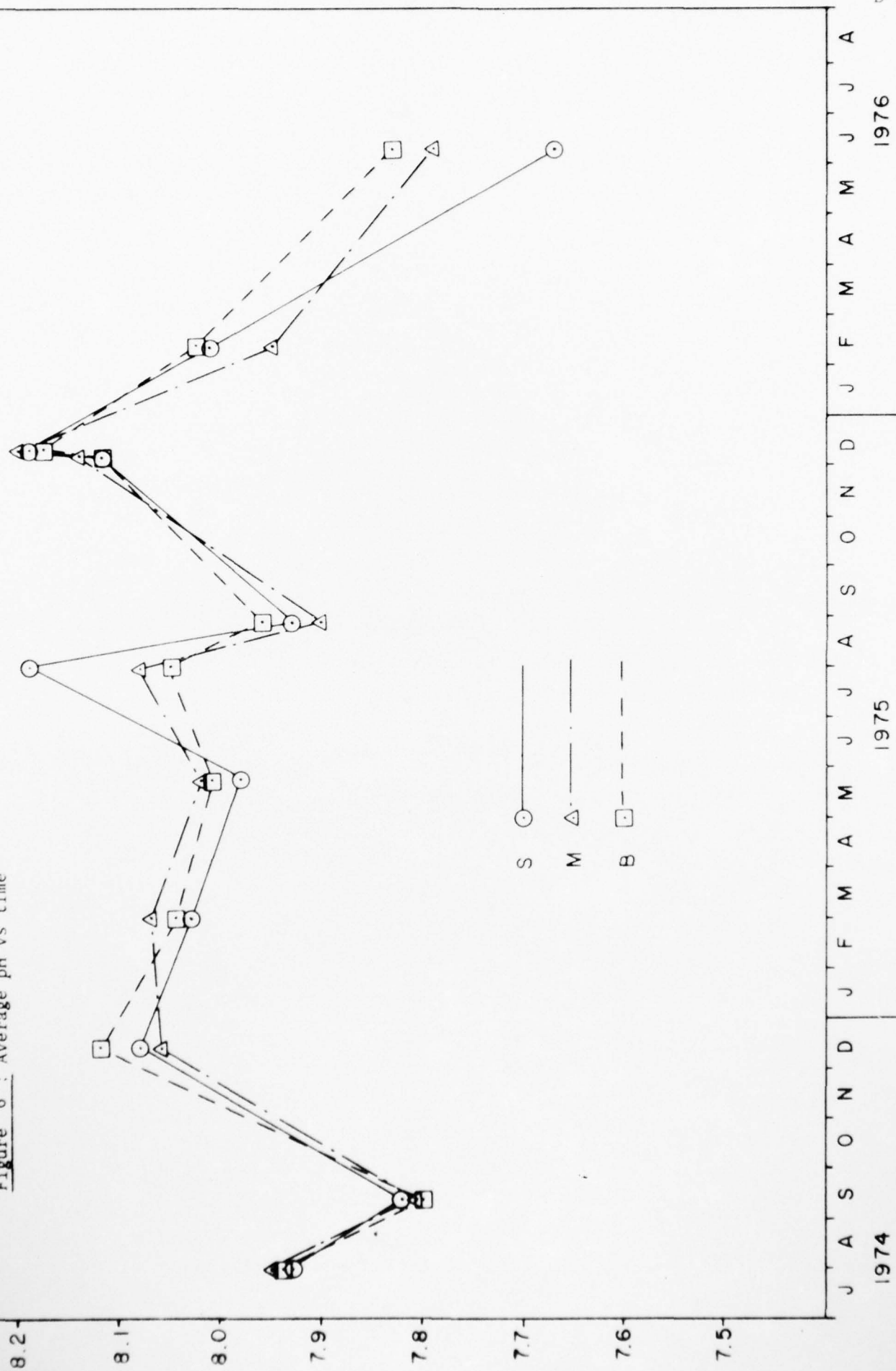


Figure 7 : Average volatile solids vs time (% total suspended solids)

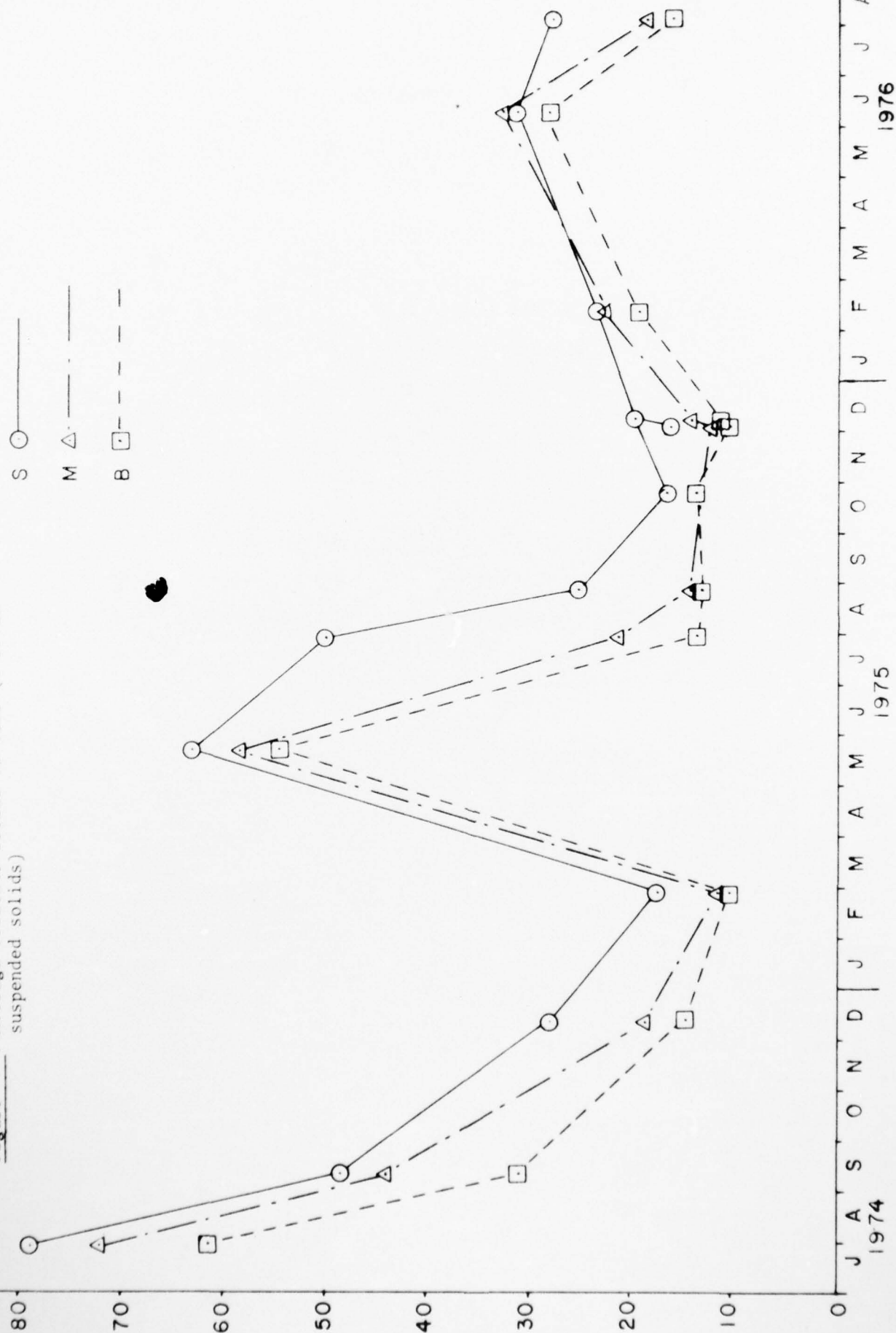
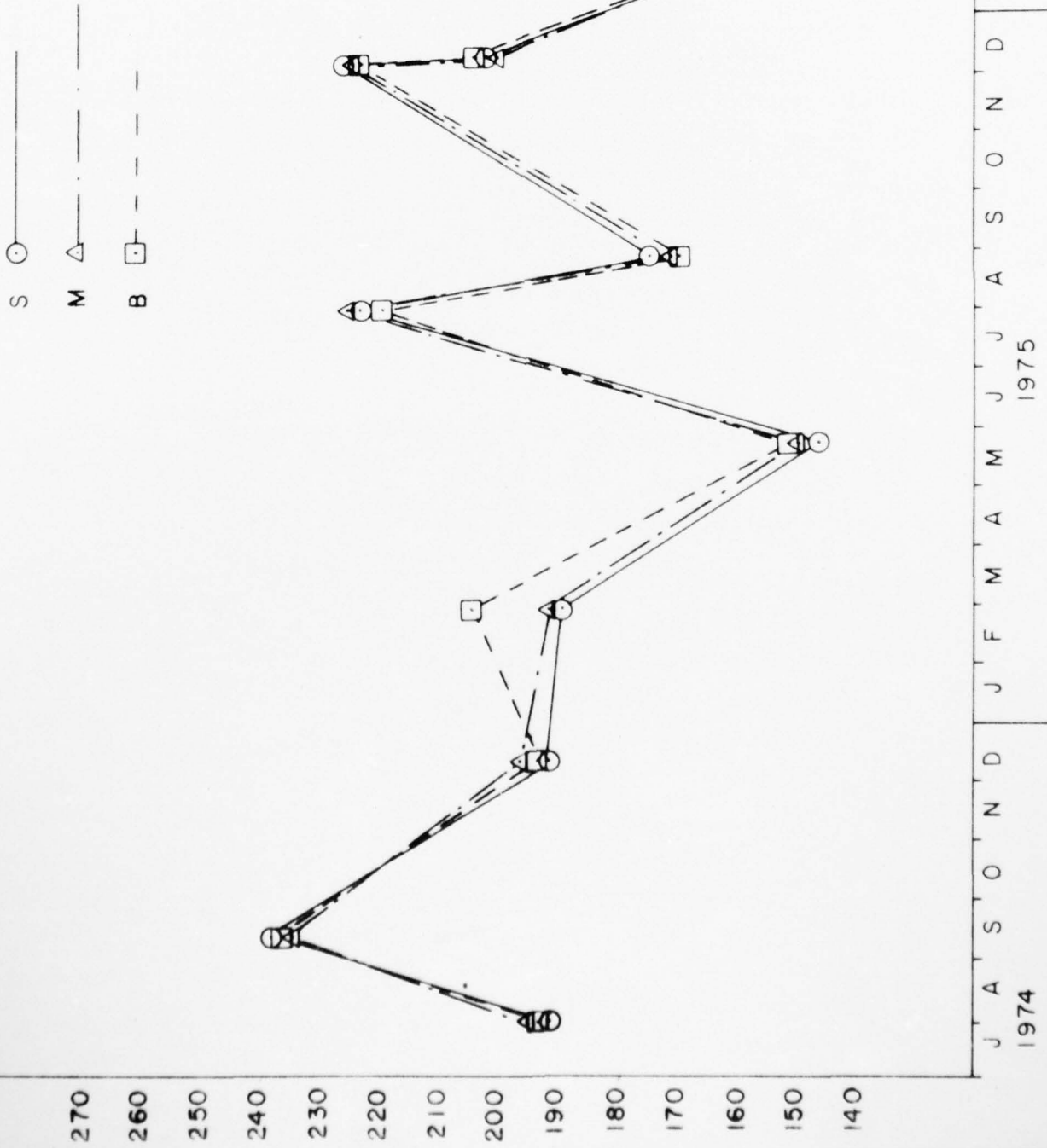


Figure 8 : Average Eh vs time (MV)



always, higher as the water depth increases. The seasonal changes of turbidity, however, do not always follow the changes in suspended solids. This must reflect the fact that the particle size and nature of suspended matter have large effects upon the optical quality of the water while suspended solids depend only on mass retained by the filter.

Volatile solids also show a seasonal variation expressed as percent total solids, which are the reverse of that shown by suspended solids. This may be due to the elevated phytoplankton levels which are present in the summer months.

The above results indicate that there is no noticeable effect on the water quality due to the dump area.

There were exceptions to the generalization that the values of the parameters showed no significant variation over the sampling area. On Sept. 11, 1974 dissolved oxygen values west of the Central Buoy ranged between 84 and 89% of saturation while at and to the east of this buoy the concentrations were between 71 and 83% of saturation. Similar variations were not observed at other times, and other parameters did not vary in an analogous fashion at that date.

Drogue Studies of Resuspension

Experiments were performed to study the effect upon the value of the parameters in the water column as the water passes over the dump area. Drogues, at two levels, were placed in the water when or before it

passed over the dump area. Samples were drawn at the drogues at a number of time intervals up to a few hours. Again, it was observed that the values of the parameters in the water were unaffected by the passage of the water over the dump area. Hence, no resuspension of spoil was shown by these experiments. These studies were performed on various dates so that all flow conditions were sampled, with emphasis on maximum current velocities.

Post-Dump Studies

In Sept. 1974 and March 1975, drogues were placed in the water at two depths immediately after a dump occurred at the spoil dumping location. Samples of the water at the drogues were then taken as a function of time. This study permits the determination of the time needed for the parameters in the spoil plume to return to background.

The water column parameters most affected by a dump were dissolved oxygen, suspended solids and turbidity. In the first of these experiments the dissolved oxygen content in the bottom water dropped to about 48% of saturation and returned to the ambient 84% within 40 minutes. The surface and middle waters were hardly affected. In the March experiment, the bottom waters took less than 10 min to return from 65% saturation to the ambient 96% saturation, and again, surface waters were unchanged.

In the Sept. study, suspended solids in surface waters returned to ambience within 10 mins of spoil while bottom waters following the surface drogue took about two hours to reach ambience. The March experiment showed

that the effects of the dump were completely unobservable after about 30 min. At the time of the dump suspended solid values rose from 10 to 100 times the ambient values.

Turbidity samples, taken at the same time as the suspended solid samples, showed like effects due to the dumping. In the Sept. (1974) study, during the anchored portion of the study, turbidity increased approximately 150 times and returned to the pre-dump condition within 40 minutes. In a drogue experiment performed the same day, bottom waters in the vicinity of the surface drogue, required 1-1/2 hours to return to ambience following a spoiling. On March 6, 1975, during a study where both surface and bottom drogues were followed, no elevated turbidity values were detected after the initial sampling in any of the water samples. This, in conjunction with suspended solid data reviewed earlier, indicated that the spoil material deposition rate is rapid.

The pH and Eh values were apparently unaffected by the dumping as determined by these drogue studies.

On May 21, 1975, stations downstream of the dump site on an ebb tide (eastern stations) and one each north, south and west of the site, were monitored following a spoil dumping event. Perturbations of the ambient values of some parameters were observed only at the closest downstream station (E1) at approximately the time predicted from current meter readings (1 hr). pH values decreased about 0.3 units at the surface and

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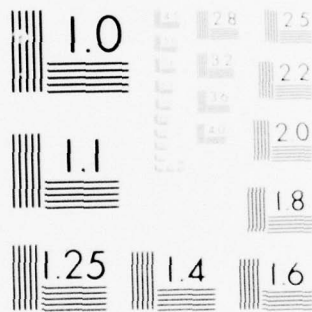
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suspended solids were higher than ambient. In bottom waters, suspended solids rose about seven times ambient. These values returned to ambient within about a half hour. Dissolved oxygen values showed no significant change from ambient as expected from the drogue studies described above in which dissolved oxygen values were found to return to ambient well within one hour.

Sediments

The sediments at the dump site and its environs are of great interest with regard to the question of whether the spoils, or any of the components of the spoils, remain at the dump site or tend to spread out beyond the one square mile designated as the dump site. The parameters investigated in the sediments were the metals, cadmium, copper, iron, nickel, lead, zinc and mercury, Kjeldahl nitrogen, total phosphorus, and chemical oxygen demand.

The analytical methods used have been described above. For the last two sample periods, the sample size has been increased from 1 gram to 10 grams. This change affords an approximately 10 fold decrease in detection limits.

Table 8 lists the minimum detectable values obtained on these samples. The minimum detectable values listed are estimated by taking .002 absorbance units as the lowest reliable value to be obtained from flame spectrophotometric analysis. The actual minimum concentration detected depends upon the sensitivity of the particular metal being analyzed. Table 8 also compares

Table 8

Minimum Detectable Values

Element	Cd	Cu	Fe	Ni	Pb	Zn	Hg ¹
Minimum detectable for 1 gram sample	2ppm	4ppm	6ppm	8ppm	20ppm	.9ppm	.01ppm
Minimum detectable for 10 gram sample	0.2ppm	0.4ppm	0.6ppm	1ppm	2ppm	.09ppm	.003ppm ²
Minimum detectable reported by Dr. Feng	0.3ppm	0.3ppm	-	-	1.7ppm	.24ppm	.1-.2ppb

¹Cold vapor technique

²Based on a 2 gram sample

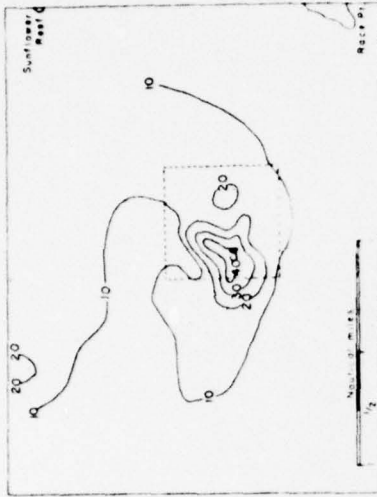
this value with those reported by Feng (1976).

Original data for September, October, 1975 and January, February, 1976 have not previously been reported and are therefore presented in appendix 1.

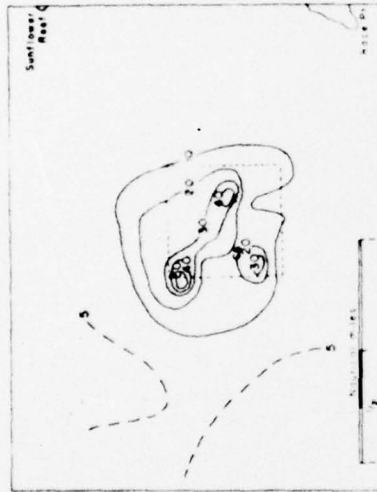
Data for copper, iron, zinc, nitrogen, phosphorus, and chemical oxygen demand are summarized in the contour diagrams, Figures 9 to 14 . Data for nickel, mercury, cadmium and lead were not contoured due to the paucity of data above minimum detectable or the small random variations versus location. The mean concentration of these parameters within the dumpsite area were compared with those outside the dumpsite. The results of this comparison are given in Table 9 . For these calculations, Stations E4, A5 and F8 were included as inner stations along with A7, A8, C5, C6, C7, C8, E5, E7, E8, F4, F5, and F7 which are within the dump area (Fig. 2). The differences between the results of the inner and outer stations were tested for significance by the t-test at 95% confidence limits (Youden, 1951). The results of these tests are shown in Table 9.

Table 9 shows that except for Kjeldahl nitrogen, total phosphorus, and chemical oxygen demand, the parameters had the same values inside and outside the dump area during the first sampling period, which was the pre-dump period. The copper and iron content inside the dump area are significantly higher than outside for all other sampling times. The zinc content shows no significant variation inside and outside the dump area while lead, nickel, and cadmium are higher inside the dump area on only a few occasions. Mercury concentrations inside the dump area are not significantly

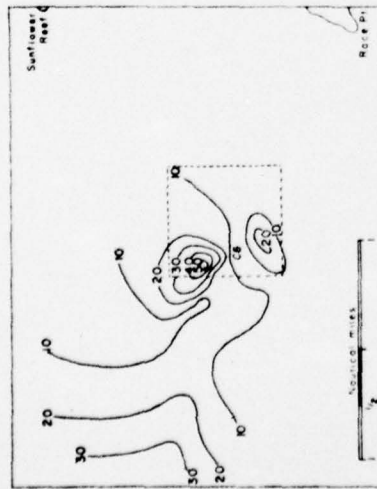
January, February 1975



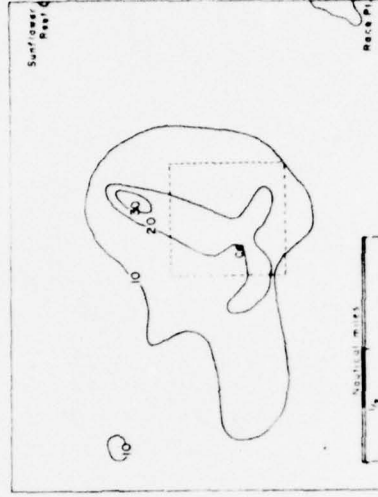
September, October 1974



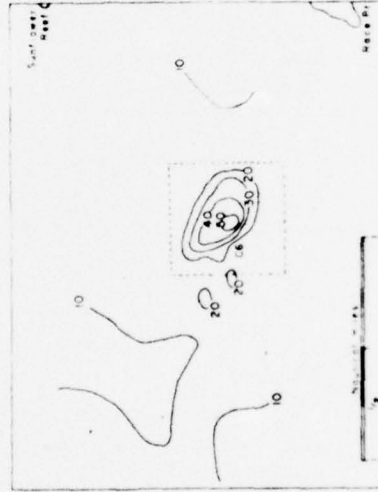
June, July 1974



September, October 1975



June, July, 1975



May 1975

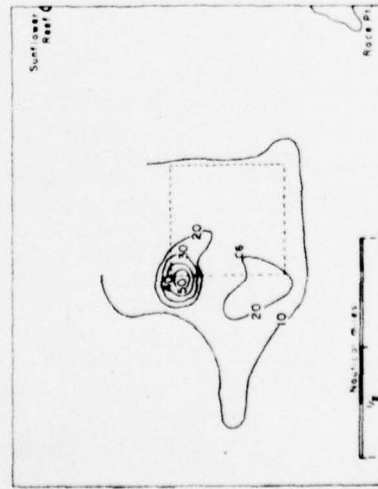
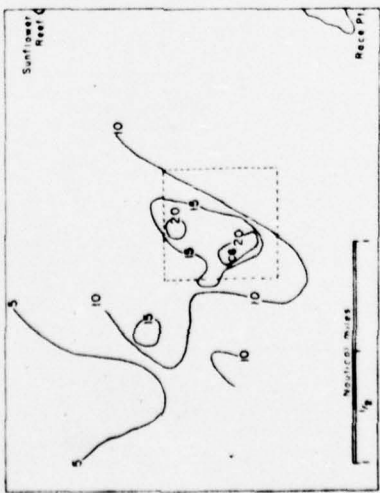
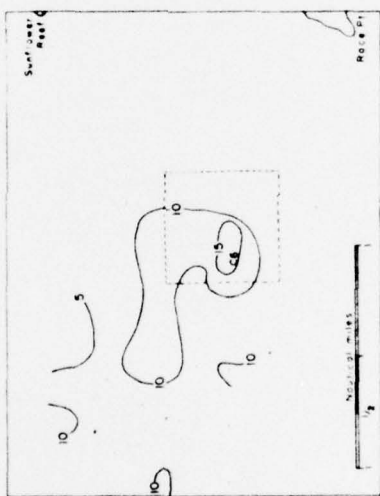
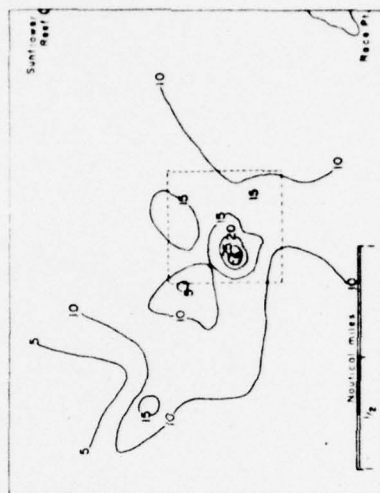
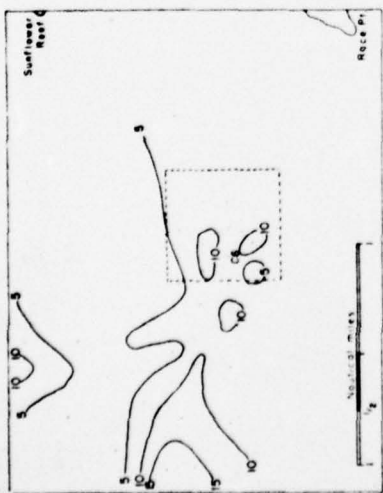
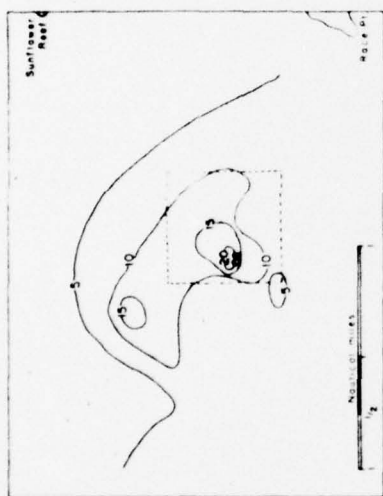
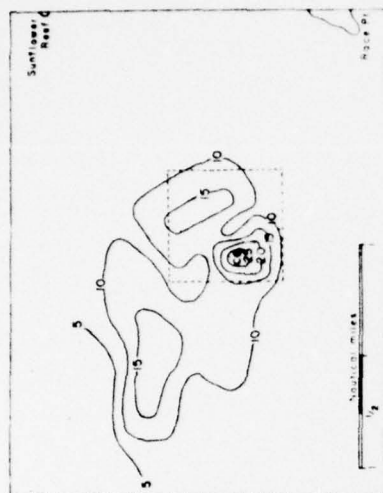


Figure 9 . New London Dumpsite Sediment Contours
Copper in ppm

January, February 1975

September, October 1974

June, July 1974



September, October 1975

June 1975

April 1975

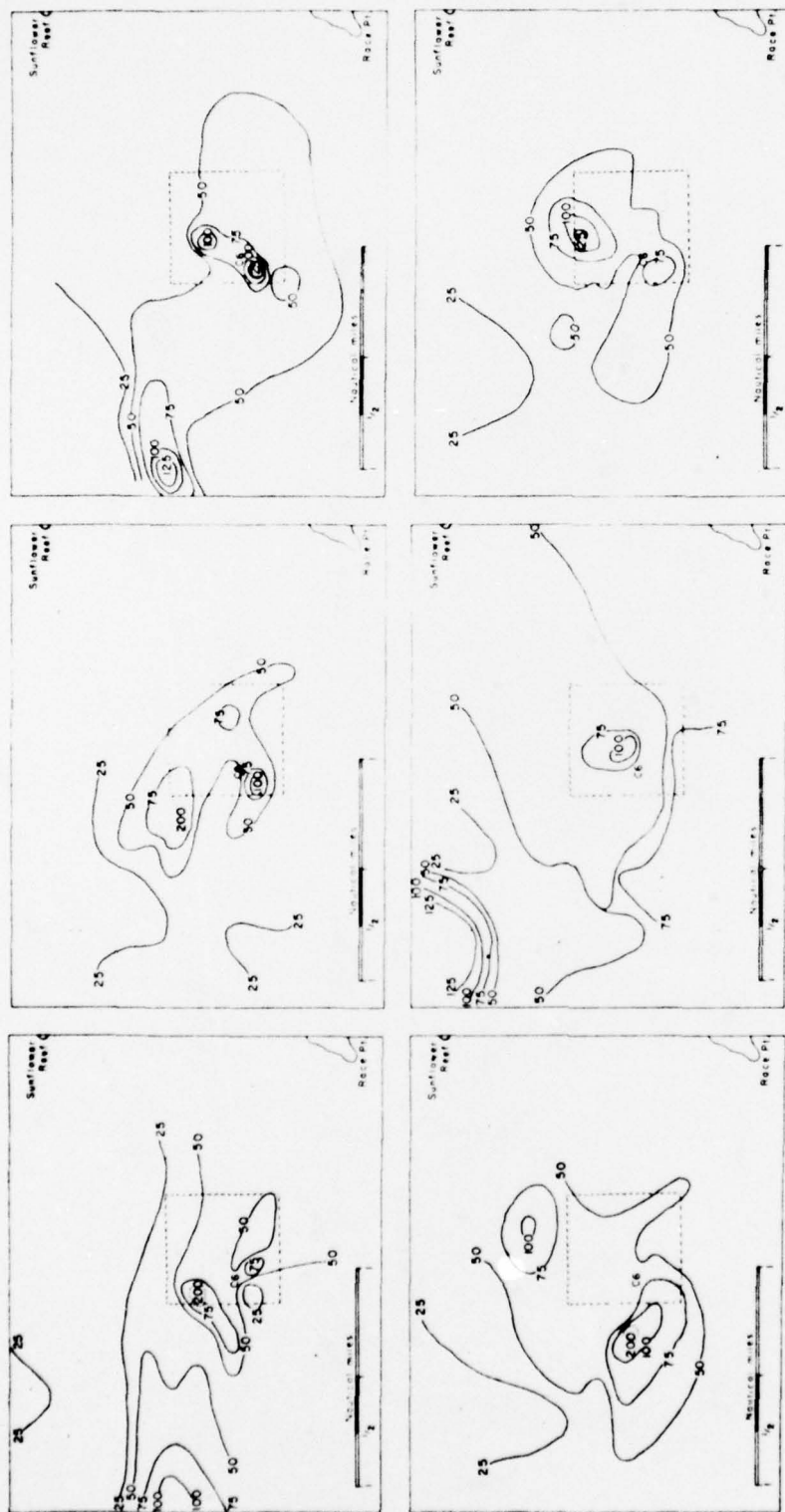
Figure 10. New London Dumpsite Sediment Contours

Iron in ppm x 10³

June, July 1974

September, October 1974

January, February 1975



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May 1975

June, July 1975

September, October 1975

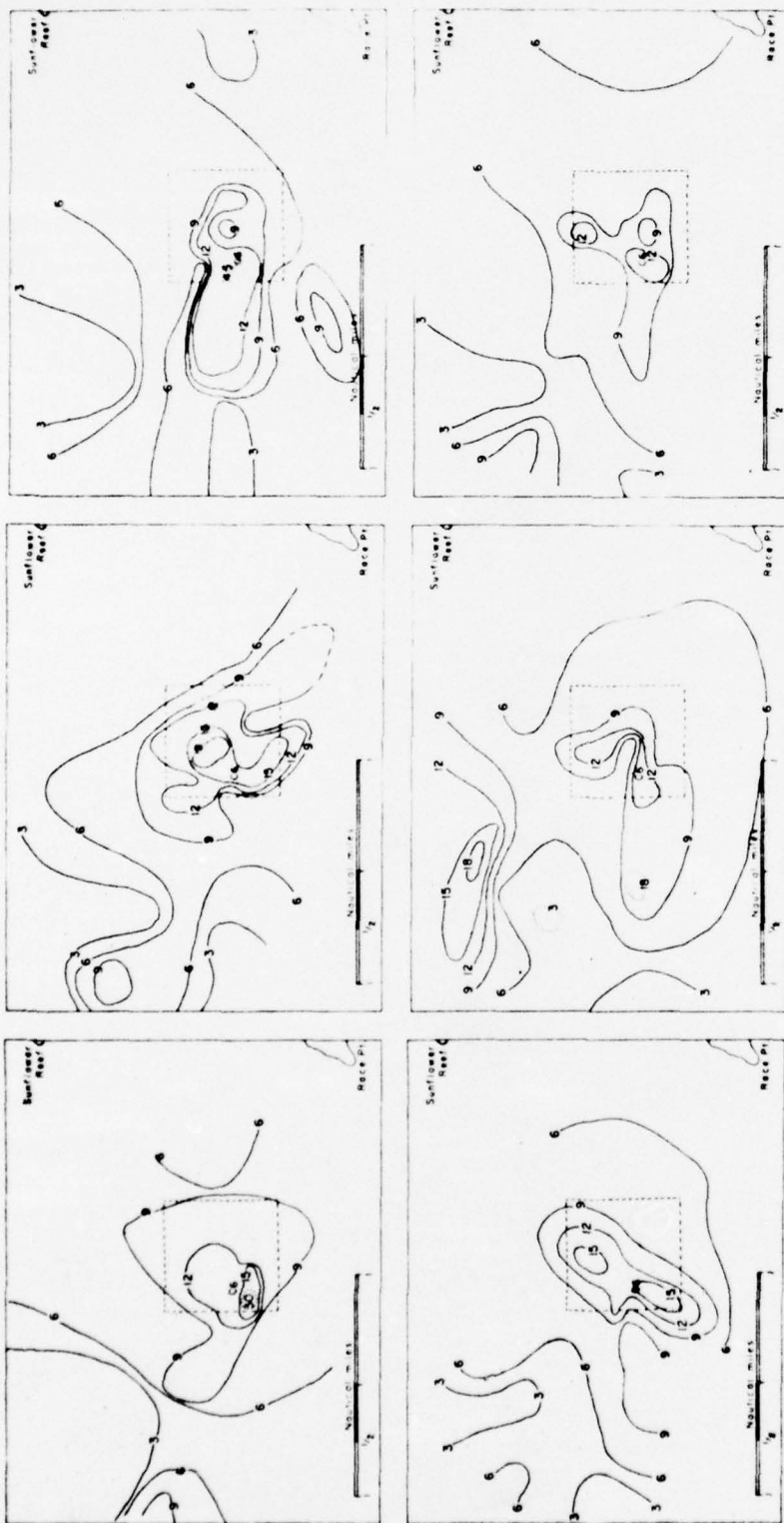
Figure 11. New London Dumpsite Sediment Contours

Zinc in ppm

January, February 1975

September, October 1974

June, July 1974



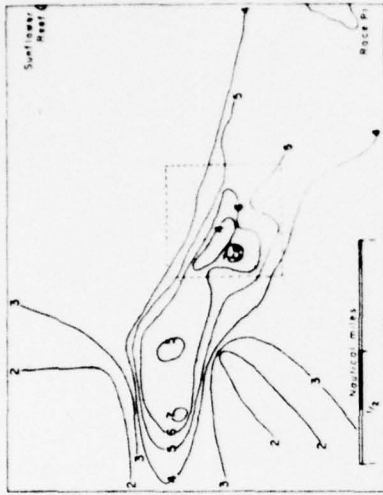
September, October 1975

June 1975

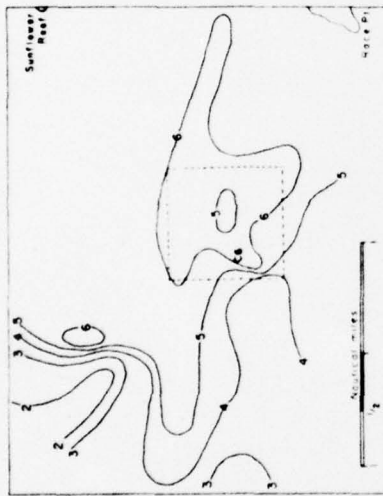
April 1975

Figure 12. New London Dumpsite Sediment Contours
Kjeldahl nitrogen in $\mu\text{g/g} \times 10^2$

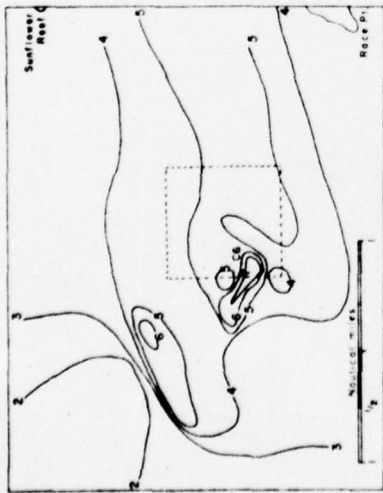
January, February 1975



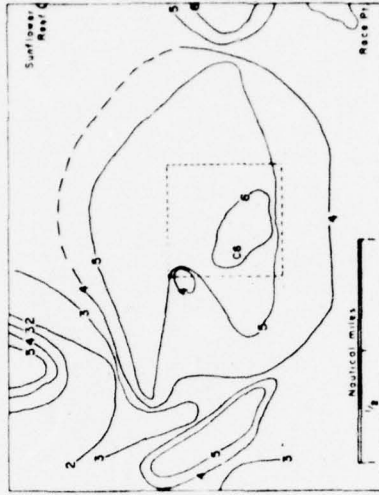
September, October 1974



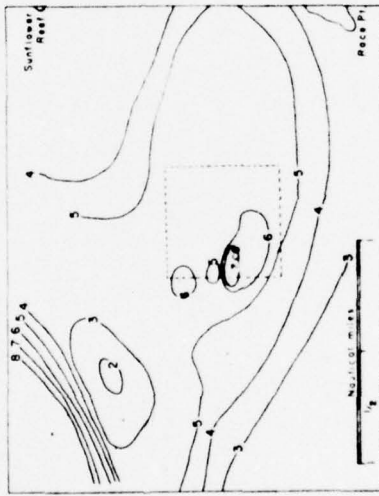
June, July 1974



September, October 1975



June, 1975



April 1975

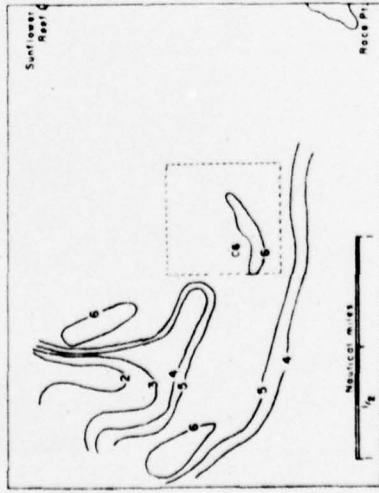
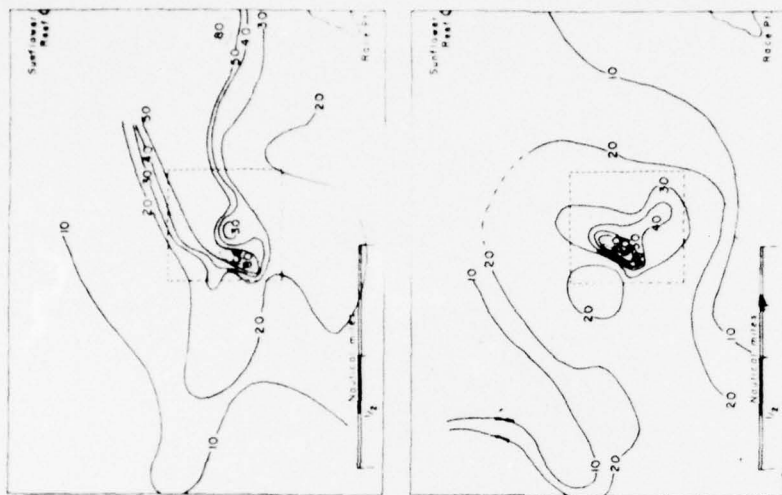


Figure 13. New London Dumpsite Sediment Contours

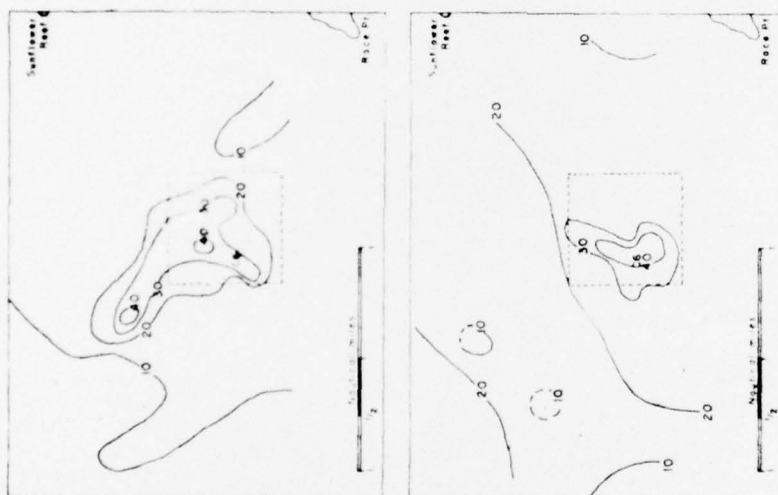
Total phosphorus in $\mu\text{g/g} \times 10^2$

January, February 1975



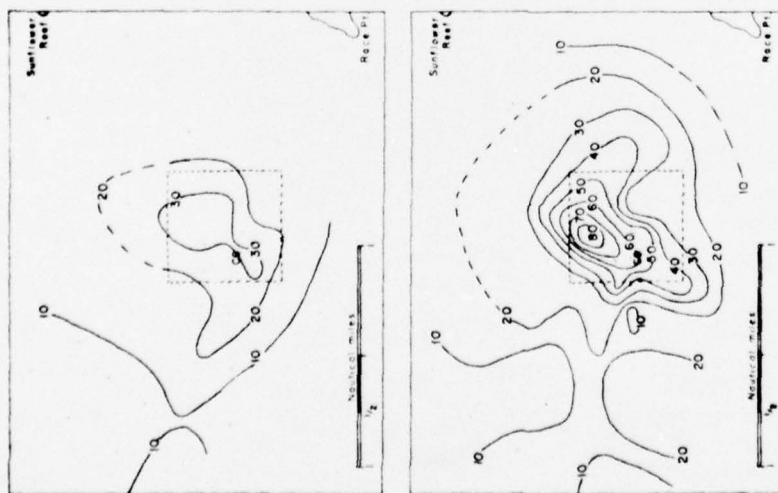
September, October 1975

September, October 1974



June 1975

June, July 1974



April 1975

Figure 14. New London Dumpsite Sediment Contours
Chemical Oxygen Demand in $\text{mg/Kg} \times 10^3$

Table 9
Means of Inner Versus Outer Stations
for Heavy Metals in Sediments
Cu, ppm

Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Different at 95% Confidence Level
June, July 1974	12.5	9.5	13	13.7	12.3	14	No
Sept, Oct 1974	6.9	4.1	22	22.3	13.1	15	Yes
Jan, Feb 1975	9.2	5.6	21	21.1	12.6	15	Yes
April 1975	8.4	5.9	21	19.9	10.1	15	Yes
June 1975	12.5	6.1	19	23.7	13.1	13	Yes
Sept, Oct 1975	8.5	4.7	21	18.3	6.4	15	Yes

Fe, ppm $\times 10^3$

June, July 1974	7.2	4.2	18	7.8	2.6	15	No
Sept, Oct 1974	6.5	3.3	22	10.4	4.9	15	Yes
Jan, Feb 1975	9.4	4.8	22	13.5	6.3	15	Yes
April 1975	8.0	3.8	22	14.7	4.9	15	Yes
June 1975	8.9	3.4	19	12.8	3.4	13	Yes
Sept, Oct 1975	8.4	3.6	21	13.2	4.9	15	Yes

Zn, ppm

June, July 1974	44.4	30.7	18	60.6	43.1	15	No
Sept, Oct 1974	33.5	15.0	22	67.8	45.7	15	Yes
Jan, Feb 1975	49.3	26.4	22	64.4	26.8	15	No
April 1975	50.5	38.1	22	59.4	14.2	15	No
June 1975	51.5	24.0	19	67.1	20.3	13	No
Sept, Oct 1975	40.0	14.7	21	60.3	24.1	15	Yes

Table 9 (cont.)

Hg, ppm							
Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Different at 95% Confidence Level
June, July 1974	.056	.094	13	.118	.069	14	No
Sept, Oct 1974	.573	.884	21	.803	.719	15	No
Jan, Feb 1975	.099	.079	22	.565	.766	14	Yes
April 1975	.057	.035	23	.044	.037	15	No
June 1975	.045	.023	17	.027	.021	13	Yes
Sept, Oct 1975	.048	.067	20	.012	.019	15	No

Kjeldahl Nitrogen, $\mu\text{g/g} \times 10^2$							
June, July 1974	5.9	3.4	18	12.2	5.9	15	Yes
Sept, Oct 1974	6.0	3.0	21	13.9	4.3	15	Yes
Jan, Feb 1975	10.3	7.7	22	20.4	11.1	15	Yes
April 1975	5.4	2.7	22	11.7	3.3	15	Yes
June 1975	6.8	4.3	19	9.9	2.0	13	Yes
Sept, Oct 1975	5.4	2.6	21	10.1	3.0	15	Yes

Total Phosphorus, $\mu\text{g/g} \times 10^2$							
June, July 1974	3.8	1.8	20	4.4	1.6	15	Yes
Sept, Oct 1974	4.8	1.4	22	5.6	0.9	15	Yes
Jan, Feb 1975	4.0	1.8	22	5.3	1.1	15	Yes
April 1975	4.6	1.6	22	5.7	0.4	15	Yes
June 1975	4.8	1.7	19	5.7	1.0	13	No
Sept, Oct 1975	4.5	1.3	21	5.7	0.9	15	Yes

Table 9 (cont.)

Chemical Oxygen Demand, mg/Kg x 10³

Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Different at 95% Confidence Level
June, July 1974	11.2	7.7	18	25.2	7.4	15	Yes
Sept, Oct 1974	11.4	8.9	22	25.5	10.1	15	Yes
Jan, Feb 1975	15.4	8.5	22	37.3	19.1	15	Yes
April 1975	16.9	10.1	22	46.5	19.4	15	Yes
June 1975	15.7	8.8	19	32.9	8.8	13	Yes
Sept, Oct 1975	16.6	8.3	21	35.9	16.9	14	Yes

Pb, ppm

June, July 1974	32.0	11.0	9	29.0	10.0	13	No
Oct 1974	23.0	10.0	3	26.0	7.0	11	No
Jan 1975	25.0	6.0	4	27.0	15.0	7	No
May 1975	30.0	0	2	<20.0		0	No
June 1975	29.0		1	35.0	8.0	5	No
Sept 1975	9.6	6.7	12	20.2	17.6	11	Yes
Jan 1976	17.8	7.0	8	13.7	3.6	3	No

Cd, ppm

June, July 1974	<2.0		0	<2.0	0	0
Oct 1974	9.0	0	1	4.0	0	1
Jan 1975	3.0		1	3.0		1
May 1975	<2.0		0	<2.0		0
June 1975	11.0		1	4.5		2
Sept 1975	.46	.82	8	.42	.22	9
Jan 1976	.30		1	.08		1

Table 9 (cont.)

Ni, ppb							
Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Different at 95% Confidence Level
June, July 1974	7.3	3.5	7	10.0	4.7	8	No
Oct 1974	5.7	2.1	16	9.0	3.3	15	Yes
Jan 1975	12.1	9.6	14	16.0	4.4	13	No
May 1975	13.9	1.7	9	16.5	2.9	14	Yes
June 1975	11.0	1.7	3	14.8	4.4	4	No
Sept 1975	7.8	1.3	16	13.2	5.6	15	Yes
Jan 1976	6.3	3.6	7	8.4	1.5	3	No

higher than those outside except for January and February, 1975. After an initial increase in September, 1974, the mercury content in the dump area decreased. Therefore, the changes in mercury content of the dump area do not appear to be attributable to spoil dumping. Chemical oxygen demand, Kjeldahl nitrogen, and total phosphorus were always significantly higher in the dump site. The fact that the pre-dump period is no different than the ones after the dumping commenced must be due to the prior use of this site.

The contours in Figures 9-14 show that there is no general trend such as spreading of the dredged material out of the dump area.

No variations with respect to time are apparent except for the initial increase in copper and iron concentrations. This suggests that the increase in concentration of these metals is due to the dumping of dredge spoil.

It is interesting to note that an earlier study of the sediments in this area (NAVOCEANO, 1973) report a copper concentration of about 10 ppm for their phases 1 and 2 which is similar to the "outer" copper concentrations observed in this study. Their phase 3 copper results were about 31 ppm which is roughly 50% higher than the "inner" values found here. This earlier study reported zinc concentrations ranging from about 60 to 80 ppm which is the order of magnitude of the zinc content found in the present study. Cadmium results are also similar in both studies. Mercury was reported as less than 1 ppm in the NAVOCEANO study which is in agreement with the results

found here. Lead, however, ranged from about 100 to 200 ppm in the earlier study while the present study reports lead content in the 20-30 ppm range. It must be pointed out that the NAVOCEANO study used a total digestion treatment of the samples. Lead in the Thames River sediments (Feng, 1976) ranged from the order of 20 to 150 ppm.

Benthic Organisms

Effects of spoil dumping on benthic organisms is an important part of the total environmental impact of the dump site. Benthic organisms were collected by the Sandy Hook Laboratory. These organisms were analyzed for iron, zinc, copper, cadmium, nickel, lead and mercury, Kjeldahl nitrogen and total phosphorus at NYOSL as described above.

A complete reporting of results of the determinations made on the benthic animals is given in Table 10 and the quarterly reports. Although many species were present in the samples, only *Pitar* and *Mercenaria* were found to be abundant at all times and at most stations. Unfortunately, the identifications of the samples collected in the June, October, 1974, and the January and April, 1975 cruises are in doubt. The question remains whether *Pitar* and *Mercenaria* were confused in these samples. It is interesting, however, to compare the data for metals in *Pitar* and *Mercenaria* in Table 10 in which the results are averaged. In the Sept., 1975 and May, 1976 samples, which were identified by Sandy Hook biologists, the metal content, except for cadmium, are virtually the same in the two species. *Mercenaria* and *Pitar* taken from the river (Feng, 1976) showed

Table 10
Heavy Metals in Benthics
Values in ppm (dry wt.)

Species	Station	Number of Individuals	Cd	Cu	Fex10 ²	Ni	Pb	Zn	Hg
<i>Mercenaria mercenaria</i>	A8	1	0.8	20	1.6	17	2	260	.15
	A8	1	<0.1	14	5.2	2	4	150	.16
	G1	1	0.7	20	2.3	14	2	190	↑
	G1	1	0.5	14	8.8	8	5	150	
	G1	1	0.7	17	3.6	10	5	300	
	G1	1	0.6	9	2.0	17	2	190	<0.3
	G1	1	1.4	17	2.3	11	3	150	
	G1	1	0.6	13	7.3	6	2	140	
	G1	1	2.0	12	1.8	15	3	140	
	G1	1	0.9	14	1.4	8	2	110	
	G1	1	1.3	6	4.0	18	3	310	
	G1	1	2.0	23	7.3	3	6	180	
	G1	1	0.9	20	0.9	6	3	190	
	G1	1	0.8	13	0.9	17	3	180	
	G1	1	0.6	11	4.4	1	<2	69	
	F3	1	1.0	23	0.6	14	<2	67	↑
	F3	1	0.9	18	1.5	9	3	180	0.4
	C9	1	1.8	33	5.8	9	5	130	↑
	A10	1	1.8	21	1.1	11	6	300	
	A10	1	1.1	10	5.3	10	2	120	
	A10	1	0.4	13	1.2	15	2	180	
	A10	1	1.4	7	5.4	14	2	260	<0.3
	A10	1	0.7	12	1.2	13	3	160	
	A10	1	1.7	6	3.5	8	3	130	
	A10	1	1.3	15	0.8	12	3	130	
	A10	1	0.7	8	1.2	10	2	88	↑
	C8	1	1.9	9	9.2	14	2	180	0.8

Table 10 (cont.)

Species	Station	Number of Individuals	Cd	Cu	Fex102	Ni	Pb	Zn	Hg
<i>Pitar morrhuana</i>	A3	2	0.5	9	5.6	4	5	580	IS
	F8	4	1.0	13	4.4	9	5	310	0.12
	F3	3	0.4	16	9.7	2	5	160	IS
	A9	2	0.6	14	9.1	1	8	140	0.12
	C7	1	IS	IS	IS	IS	IS	IS	0.12
	C9	2	2.0	16	5.1	10	10	280	0.14
	A8	2	0.5	14	3.2	3	<2	81	<0.08
	C6	1	1.8	15	3.6	<1	<2	140	IS
	C8	4	0.1	13	3.0	1	2	90	0.12
	C8	4	1.3	14	4.3	7	3	140	0.10
	A9	1	3.0	12	6.2	8	12	150	0.23
	A9	1	2.2	13	4.6	5	12	110	0.15
	A9	1	1.6	13	2.2	5	6	100	<0.08
	A9	1	1.4	11	4.0	2	5	260	0.08
	A9	1	1.3	13	4.3	3	5	300	0.37
	A9	1	0.1	10	1.7	<1	4	120	0.35
	A9	1	0.8	14	3.1	2	5	140	0.83
	A9	1	0.3	9	0.7	<1	<2	68	0.16
	A9	1	1.5	16	5.6	7	6	150	0.33
	A9	1	2.2	14	8.1	6	7	170	0.26
	A9	1	3.0	13	4.3	2	10	150	0.34
	A9	1	1.0	13	7.0	<1	6	140	<0.08
	A9	1	<0.1	20	14.5	<1	<2	87	0.20
	A9	1	<0.1	17	2.0	<1	5	360	0.49
	A9	1	1.3	14	3.3	2	10	190	0.14
	F3	3	0.6	16	4.8	4	2	130	0.10
	F3	3	1.0	18	5.2	7	7	130	0.09
	F3	2	0.6	14	2.7	2	6	120	<0.08
	F3	2	0.7	17	3.8	4	4	66	0.10
	F3	1	<0.1	9	1.8	<1	<2	170	0.19
	G1	1	2.8	15	5.7	3	10	110	IS
	G1	1	0.8	13	6.8	5	12	60	IS
	G1	1	0.9	12	3.5	4	5	340	↑

IS = Insufficient Sample

Table 10 (cont.)

Species	Station	Number of Individuals	Cd	Cu	Fex102	Ni	Pb	Zn	Hg
<i>Pitar morrhuana</i> (cont.)	G1	1	1.6	11	4.8	9	10	220	IS
	G1	1	1.8	12	7.7	<1	10	240	
	G1	1	1.4	12	7.9	<1	9	120	
	G1	1	1.3	12	1.2	2	9	210	
	G1	1	0.9	15	4.7	5	9	120	
	G1	1	0.5	12	2.0	2	8	170	
	G1	1	0.9	15	5.1	2	10	150	
	G1	1	0.3	15	2.5	32	9	140	
	G1	1	0.8	30	6.6	47	11	180	
	G1	1	0.4	12	2.0	9	6	180	
	G1	1	0.7	19	5.5	5	5	130	
	G1	1	0.9	11	3.9	<1	11	250	
	C9	1	0.2	16	3.2	2	2	340	
	C9	1	0.7	17	2.7	3	7	200	
	C9	1	1.8	15	9.0	<1	16	130	
	C9	1	2.8	20	2.1	<1	10	110	
	C9	1	1.1	17	10.8	<1	8	190	
	C9	1	1.1	17	3.3	32	7	110	
	C9	1	3.5	16	3.7	6	6	450	
	C9	1	0.4	15	2.7	4	11	120	
	C9	1	2.0	18	6.7	2	9	190	
	A10	1	5.3	14	10.4	7	15	120	<0.3
	A3	1	0.8	18	4.2	12	8	160	<0.3
<i>Astarte undata</i>	A3	1	<		IS				0.32
	F3	2	<		IS				0.16
	A9	3	0.6	23	8.5	20	4	150	IS
	C9	2	<		IS				0.08
	C8	1	<		IS				0.73
	A9	3	9.2	23	11.3	<1	2	70	0.57

IS = Insufficient Sample

Table 10 (cont.)

Species	Station	Number of Individuals	Cd	Cu	Fex10 ²	Ni	Pb	Zn	Hg
<i>Astarte undata</i> (cont.)	A9	3	<0.1	19	5.9	<1	<2	49	0.12
	A9	4	1.4	17	6.6	3	5	110	0.20
	G1	4	1.1	45	4.9	<1	3	49	IS
	G1	4	0.2	24	9.9	<1	2	69	IS
<i>Venericardia borealis</i>	C3	2	3.4	26	8.8	<1	10	37	IS
	F3	4	1.8	21	8.4	<1	5	73	0.19
	F3	4	5.1	38	5.9	<1	7	140	0.53
	F3	5	2.6	30	3.9	<1	5	210	0.20
	G1	5	3.5	36	5.2	<1	7	50	IS
<i>Glycyra</i> sp.	A8	1	1.0	27	5.8	<1	<2	360	0.09
	F3	1	2.2	22	4.6	<1	6	150	0.3
	A10	1	2.2	8	3.8	<1	<2	250	<0.5
	A3	1	1.8	37	7.0	7	5	120	IS
<i>Lunatia heros</i>	F3	1	1.7	20	3.8	<1	<2	140	1.4
	G1	1	1.0	21	2.5	3	<2	81	<0.5
<i>Pagurus pollicaris</i>	A10	1	1.0	340	9.9	1	13	79	<0.3
	A10	1	<0.1	47	1.8	<1	4	30	
	A10	1	1.0	200	12.4	2	5	8.2	
	A10	1	0.5	220	9.4	3	10	99	
	A10	1	<0.1	15	1.4	7	3	-	
	A3	1	0.6	140	3.1	2	12	97	
	A3	1	<0.1	58	8.4	4	14	120	
	A3	1	<0.1	270	6.4	3	10	130	
	A3	1	1.1	200	8.0	3	16	76	
	A3	1	1.2	230	4.3	5	11	84	
	A3	1	1.0	150	7.9	6	13	140	
	A3	1	0.7	53	6.0	7	13	22	<0.3
<i>Artica islandica</i>	A3	1	2.2	11	2.9	12	8	200	<0.3
	C9	1	1.6	9	7.8	6	10	100	<0.3

IS = Insufficient Sample

a similar relationship in their copper, nickel and mercury content, while the zinc content was slightly higher in *Pitar*. Cadmium again was appreciably higher in *Pitar* than in *Mercenaria*. In view of the similarity of metal content of *Pitar* and *Mercenaria*, and the lack of any trend in the data in Table 10 with respect to time, it may be concluded that the metal content of *Mercenaria* did not increase over the period covered in these studies.

Table 11 compares data for animals found inside the one square mile dump area with data for animals found outside the area. The specific stations inside the area are listed above in the "sediment" section. Again, only *Mercenaria* and *Pitar* were found in sufficient quantities for tabulation. With few exceptions, the differences between the data for the "inner" animals and the "outer" animals are insignificant. The few differences observed were small.

There is a similarity between the results reported here and the metal content of *Mercenaria* and *Pitar* found in the Thames River (Feng, 1976). *Mercenaria* in the river had zinc contents ranging from 170 ppm to 5300 ppm, copper from about 16 ppm to about 29 ppm, cadmium from 0.7 ppm to 2 ppm, nickel from 6 ppm to 13 ppm, and mercury from 0.2 ppm to 0.4 ppm. *Pitar* in the river had a zinc range of about 360 ppm to about 460 ppm, copper of 10 ppm to 20 ppm, cadmium of 3 ppm to 4 ppm, nickel of 5 ppm to 10 ppm, and mercury of 0.1 ppm to 0.2 ppm. Cadmium and to a lesser extent nickel, was appreciably higher in the organisms found in the sound but other parameters were quite similar.

Table 11
Means of Inner Versus Outer Stations
for Heavy Metals in Benthics
Pitar morrhuana

Cu

Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite Mean
June, July 1974	N.S.	-	0	N.S.	-	0	N.S.
Sept., Oct. 1974	↓	-	0	↓	-	0	↓
Jan., Feb. 1975	↓	-	0	↓	-	0	↓
April 1975	↓	-	0	↓	-	0	↓
June 1975	21.1	-	1	18.86	1.66	3	↓
Sept., Oct. 1975	17.20	4.52	2	15.55	.07	2	↓
May 1976	17.01	7.57	56	16.29	5.20	8	↓

Fe

June, July 1974	N.S.	-	-	N.S.	-	0	N.S.
Sept., Oct. 1974	↓	-	-	↓	-	0	↓
Jan., Feb. 1975	↓	-	-	↓	-	0	↓
April 1975	↓	-	-	↓	-	0	↓
June 1975	1815	-	1	1705	453	3	↓
Sept., Oct. 1975	2198	123	2	3421	2147	2	↓
May 1976	549	345	49	327	31	3	↓

Zn

June, July 1974	N.S.	-	0	N.S.	-	0	N.S.
Sept., Oct. 1974	↓	-	0	↓	-	0	↓
Jan., Feb. 1975	↓	-	0	↓	-	0	↓
April 1975	↓	-	0	↓	-	0	↓
June 1975	523	-	1	362	60	3	↓
Sept., Oct. 1975	425	156	2	200	114	2	↓
May, 1975	238	174	58	221	123	8	↓

Table 11 (cont.)

Cd

Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite Mean
June, July 1974	N.S.	-		N.S.	-		N.S.
Sept., Oct. 1974		-			-		
Jan., Feb. 1975		-			-		
April 1975	↓	-		↓	-		
June 1975	2.1	-	1	.83	.35	3	
Sept., Oct. 1975	1.56	1.88	2	1.35	1.76	2	
May 1976	1.38	1.26	59	2.04	2.49	7	↓

Ni

June, July 1974	N.S.	-		N.S.	-		N.S.
Sept., Oct. 1974		-			-		
Jan., Feb. 1975		-			-		
April 1975	↓	-		↓	-		
June 1975	25.4	-	1	10.66	4.36	3	
Sept., Oct. 1975	63.80	10.04	2	16.15	21.56	2	
May 1976	5.67	6.99	59	6.32	5.45	7	↓

Pb

June, July 1974	N.S.	-		N.S.	-		N.S.
Sept., Oct. 1974		-			-		
Jan., Feb. 1975		-			-		
April 1975	↓	-		↓	-		
June 1975	14.1	-	1	7.93	2.47	3	
Sept., Oct. 1975	24.50	4.94	2	16.10	4.38	2	
May 1976	8.89	4.62	55	5.31	4.51	7	↓

Table 11a
Means of Inner Versus Outer Stations
for Heavy Metals in Benthics
Mercenaria mercenaria

Cu									
Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite	Standard Deviation	n
June, July 1974	41.71	21.98	6	30.28	5.97	5	N.S.	-	0
Sept., Oct. 1974	11.7	-	1	18.22	9.67	4	17.8	-	1
Jan., Feb. 1975	17.5	-	1	14.55	6.01	2	N.S.	-	0
April 1975	N.S.	-	0	21.3	-	1	19.43	2.00	3
June 1975	N.S.	-	0	N.S.	-	0	N.S.	-	0
Sept., Oct. 1975	N.S.	-	0	N.S.	-	0	N.S.	-	0
May 1975	15.62	7.44	26	14.94	5.14	5	N.S.	-	0

Fe									
Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite	Standard Deviation	n
June, July 1974	619	268	6	1053	547	6	N.S.	-	0
Sept., Oct. 1974	2735	-	1	1798	467	4	1700	-	1
Jan., Feb. 1975	534	-	1	418	202	2	N.S.	-	0
April 1975	N.S.	-	0	119	-	1	765	351	3
June 1975	N.S.	-	0	N.S.	-	0	N.S.	-	0
Sept., Oct. 1975	N.S.	-	0	N.S.	-	0	N.S.	-	0
May 1976	445	263	12	348	175	4	N.S.	-	0

Table 11a (cont.)

Zn									
Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite	Standard Deviation	n
June, July 1974	251.8	32.2	6	230	46.7	6	N.S.	-	0
Sept., Oct. 1974	328	-	1	284	106	4	267	-	1
Jan., Feb. 1975	202	-	1	332	115	2	N.S.	-	0
April 1975	N.S.	-	0	266	-	1	272	42	3
June 1975	N.S.	-	0	N.S.	-	0	-	-	0
Sept., Oct. 1975	N.S.	-	0	N.S.	-	0	-	-	0
May 1976	195	98	27	208	51	4	-	-	0

Cd

Cd									
Date	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite	Standard Deviation	n
June, July 1974	<2	-	5	<2	-	6	N.S.	-	0
Sept., Oct. 1974	2.10	-	1	2.15	.54	4	2.8	-	1
Jan., Feb. 1975	.45	-	1	.96	.14	2	N.S.	-	0
April 1975	N.S.	-	0	6.6	-	1	2.13	1.89	3
June 1975	N.S.	-	0	N.S.	-	0	N.S.	-	0
Sept., Oct. 1975	N.S.	-	0	N.S.	-	0	N.S.	-	0
May 1976	1.35	1.47	31	.76	.62	6	N.S.	-	0

Table 11a (cont.)

Date	Ni						
	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite
June, July 1974	9.81	2.01	6	9.50	2.44	6	N.S.
Sept., Oct. 1974	14.8	-	1	12.90	3.43	4	11.4
Jan., Feb. 1975	16.4	-	1	13.60	3.11	2	N.S.
April 1975	N.S.	-	0	12.5	-	1	10.53
June 1975	N.S.	-	0	N.S.	-	0	N.S.
Sept., Oct. 1975	N.S.	-	0	N.S.	-	0	N.S.
May 1976	11.04	6.06	25	12.60	6.02	5	N.S.
							0

Pb

Date	Pb						
	Outer Mean	Standard Deviation	n	Inner Mean	Standard Deviation	n	Composite
June, July 1974	7.13	4.13	6	5.70	2.79	6	N.S.
Sept., Oct. 1974	27.0	-	1	16.50	6.40	4	18.0
Jan., Feb. 1975	<4	-	1	<5	-	2	N.S.
April 1975	N.S.	-	0	8.0	-	1	8.33
June 1975	N.S.	-	0	N.S.	-	0	N.S.
Sept., Oct. 1975	N.S.	-	0	N.S.	-	0	N.S.
May 1976	3.07	1.27	25	2.50	.95	5	N.S.
							0

Finally, analyses performed on Mercenaria and Pitar collected from the dumpsite area and analyzed by personnel of the Environmental Chemistry Task, Northeast Fisheries Center (NEFC), Milford, Conn., resulted in values quite similar to the present data. (Greig, personal communication) (Tables 12 and 13). Their samples were collected simultaneously with those run by NYOSL. Their data were originally given as ppm metal/wet weight; the data were then recalculated on a ppm/dry weight basis for comparison with the data in Table 10. Given the within species variation observed in heavy metal data, the averaged values are similar to NYOSL data (Table 14). Zinc in individual Mercenaria analyzed at the Milford Laboratory ranged from 80.5 to 241 ppm; copper from 11.3 to 25.7 ppm; cadmium from < 1.07 to 2.10 ppm; nickel from 6.47 to 16.9 ppm; and mercury from 0.32 to < 0.63 ppm. Composite samples of Pitar (5 to 10 animals per sample collected at specific dumpsite stations) contained zinc in a range of 113 to 166 ppm; copper from 16.7 to 23.7 ppm; cadmium from 1.43 to 2.78 ppm; nickel from 8.27 to 11.3 ppm; and mercury from 0.32 to < 0.40 ppm.

As reported in the 6th quarterly report on studies of dredging and spoil disposal at New London, the Environmental Chemistry Task, Northeast Fisheries Center (NEFC), has analyzed heavy metals in lobsters collected in pots set on a quarterly basis at stations at and extending from the disposal buoy to 0.5 n mi E of the buoy. Results of these analyses are presented in Table 15, and are com-

Table 12

Heavy metal values in Mercenaria mercenaria
 from stations in the dredge spoil disposal area;
 Metal values in ppm (wet and dry wt.)
 Data provided by R. Greig,
 Environmental Chemistry Task, NEFC

Lab.# and (Station)	19488 (A10)		19489 (A10)		19492 (G1)		19493 (G1)	
% Dry	7.97		8.74		12.57		9.06	
	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)
Ag	0.35	4.39	0.36	4.12	0.66	5.25	0.31	3.42
Cd	0.13	1.63	0.18	2.06	< 0.12	< 0.95	0.19	2.10
Cr	0.37	4.64	0.37	4.23	0.44	3.50	< 0.52	< 5.74
Cu	2.05	25.7	0.99	11.3	2.20	17.5	1.59	17.5
Hg	< 0.04	< 0.63	< 0.04	< 0.46	< 0.04	< 0.32	< 0.04	< 0.44
Ni	1.10	13.8	1.48	16.9	1.34	10.7	1.08	11.9
Pb	0.73	9.16	< 0.75	< 8.58	< 0.75	< 5.97	< 0.75	< 8.28
Zn	19.2	241.	14.7	168.	21.5	171.	7.29	80.5

Lab.# and (Station)	19494 (G1)		19495 (G1)	
% Dry	11.25		9.73	
	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)
Ag	0.48	4.27	0.49	4.93
Cd	< 0.12	< 1.07	0.19	1.95
Cr	< 0.53	< 4.71	0.37	3.80
Cu	2.65	23.6	2.50	25.7
Hg	< 0.04	< 0.36	< 0.04	< 0.41
Ni	1.11	9.86	0.63	6.47
Pb	< 0.75	< 6.67	< 0.74	< 7.6
Zn	16.6	148.	11.2	115.

Table 13

Heavy metal values in Pitar Morrhua
from stations in the dredge spoil disposal area;
Metal values in ppm (wet and dry wt.)

Lab.# and (Station) % Dry	19485 (F3)		19486 (A9)		19487 C9)		19498 (G1)	
	12.46		12.57		12.57		12.27	
	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)	Wet (ppm)	Dry (ppm)
Ag	0.37	2.97	0.37	2.94	0.78	6.21	0.43	3.50
Cd	0.23	1.85	0.18	1.43	0.35	2.78	0.31	2.53
Cr	0.40	3.21	< 0.54	< 4.30	0.41	3.26	0.77	6.28
Cu	2.95	23.7	2.10	16.7	2.86	22.7	2.73	22.2
Hg	< 0.05	< 0.40	< 0.04	< 0.32	< 0.05	< 0.40	< 0.04	< 0.33
Ni	1.03	8.27	1.09	8.67	1.42	11.3	1.15	9.45
Pb	1.47	11.8	1.43	11.4	1.51	12.0	1.24	10.1
Zn	17.1	137.	14.2	113.	18.7	149.	20.4	166.

Table 14

Averaged metal values in Mercenaria and Pitar;
Data given as parts per million (ppm) dry weight
Analysis performed by
Environmental Chemistry Task
Northeast Fisheries Center (NEFC)
and
New York Ocean Science Laboratory (NYOSL)

	<u>Mercenaria</u>		<u>Pitar</u>	
	<u>NEFC</u> (ppm)	<u>NYOSL</u> (ppm)	<u>NEFC</u> (ppm)	<u>NYOSL</u> (ppm)
Ag	4.93	-	3.91	-
Cd	1.62	1.06	2.14	1.21
Cr	4.44	-	4.26	-
Cu	20.2	14.74	21.32	14.53
Hg	0.44	0.31	0.36	0.21
Ni	11.61	10.81	9.42	5.49
Pb	7.71	3.04	11.32	7.15
Zn	153.9	171.6	141.25	177.67

Table 15

Heavy metal values in lobster, Homarus,
collected from the New London dredging spoils disposal site

Digestive Diverticula	July/Aug 1974		Jan. 1975		May/June 1975		July/Aug 1975	
	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm
Ag	8.2	30.16	9.9	36.4	8.8	32.4	7.3	23.1
Cd	3.5	12.87	3.7	13.6	3.2	11.8	4.5	14.2
Cu	775.	2850.	470.	1729.	400.	1471.	370.	1169.
Zn	36.	132.	35.	129.	27.	99.	25.	79.0
Tail Muscle								
	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm
Ag	< 0.2	< 1.1	< 0.15	< 0.8	< 0.2	< 1.1	< 0.2	< 1.1
Cd	< 0.1	< 0.5	< 0.12	< 0.6	< 0.1	< 0.5	< 0.1	< 0.5
Cu	6.3	33.4	5.4	28.6	7.2	38.1	4.8	25.4
Zn	13.	68.8	13.	68.8	20.	106.	19.	101.
Gills								
	Wet ppm	Dry ppm	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm	Wet ppm*	Dry ppm
Ag	---	---	< 0.4	< 4.3	< 0.4	< 4.3	< 0.7	< 7.5
Cd	---	---	0.47	5.0	0.3	3.2	0.4	4.3
Cu	---	---	25.	268.	19.	204.	42.	450.
Zn	---	---	15.	161.	7.5	80.	11.	118.

* Chromium, nickel, and lead analyses were conducted on these samples; all values were below the detection limits for these metals. The detection limits were:

Chromium - < 0.5 to < 0.9
Nickel - < 0.5 to < 0.8
Lead - < 0.7 to < 1.1

pared to data for lobsters from the New Haven dumpsite (Table 16). The data include ppm of heavy metals, both wet and dry weights. The New London data are based on analysis of six to ten lobsters per quarter, and for New Haven, six to eight animals. Average weights of these organisms ranged from 307 to 478 g. Variability of the metal concentrations was quite high, but the averaged data reveal no systematic changes in heavy metals in New London lobsters between July 1974 (predisposal) and July 1975 samplings. Concentrations in digestive diverticula were much higher than in tail muscles or gills. Diverticula of New Haven lobsters had higher concentrations of Ag, Cd, Cu, and Zn than did those of New London specimens. Values for tail muscles were comparable for the two areas. Neither site had detectable levels of Cr, Ni or Pb in the lobster tissues examined.

Phosphorus-nitrogen ratios are similar for the animals inside and outside the dump area. This indicates no drastic differences in the biochemistry of these animals.

Seston

Seston samples were analyzed for their cadmium, copper, iron, nickel, lead, zinc, mercury, Kjeldahl-nitrogen and total phosphorus content. Depending upon the amount of seston in the area, samples covered one or more sampling stations. Towing was always performed

Table 16

Heavy metal values in lobster, *Homarus*,
collected from the New Haven dredging spoils disposal site

Digestive Diverticula	July/Aug 1974		Jan 1975		May/June 1975		July/Aug 1975	
	Wet ppm*	Dry ppm	Wet ppm	Dry ppm	Wet ppm	Dry ppm	Wet ppm	Dry ppm
Ag	22.1	81.3	---	---	12.4	45.6	18.9	69.5
Cd	15.6	57.4	---	---	4.9	18.0	10.8	39.7
Cu	2308.	8488.	---	---	480.	1765.	906.	3531.
Zn	61.	224.	---	---	27.	99.	48.	177.
Tail Muscle								
	Wet ppm	Dry ppm	Wet ppm	Dry ppm	Wet ppm	Dry ppm	Wet ppm	Dry ppm
Ag	0.23	1.2	---	---	< 0.2	< 1.1	< 0.2	< 1.1
Cd	< 0.1	< 0.5	---	---	< 0.1	< 0.5	< 0.1	< 0.5
Cu	17.5	92.6	---	---	7.1	37.6	5.7	30.2
Zn	18.6	98.5	---	---	21.	111.	19.	101.

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* Chromium, nickel, and lead analyses were conducted on these samples; all values were below the detection limits for these metals. The detection limits were:

Chromium - < 0.5 to < 0.9
Nickel - < 0.5 to < 0.8
Lead - < 0.7 to < 1.1

along a sampling transect. The sampling stations used were those used for the water quality stations.

No relationship can be discerned between the values of the parameters and the sampling locations. This observation is consistent with what has been found in the areal survey of the water quality (see above). In addition, the variations of the parameters from one sampling period to another were random. The seston data, therefore, indicate that the parameters studied are not affected by the dumping of spoils at this site.

It should be noted that total seston concentration was not determined in this study, but, as expected, the concentration was observed to vary from period to period. It is also apparent that seston in this study includes all matter retained by the sampling net.

SUMMARY

Areal survey of the water quality showed no variation in the parameters studied with respect to location at the dump site and its environs, however, seasonal variations were observed. Seston likewise showed no differences in parameters with regard to location.

It was found that the water quality parameters returned to normal rapidly following a dumping event. The longest time observed for this return to normalcy was two hours for the bottom waters.

The sediments inside the dump area were found to be generally high-

er in iron, copper, chemical oxygen demand, total phosphorus, and Kjeldahl-nitrogen than the sediments outside the area. However, no evidence for spreading of the material outside the dump site was found.

The benthic animals examined were observed to have about the same values for the parameters studied whether they were collected inside or outside the dump area. There was no indication of significant changes in heavy metal burdens with time.

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APPENDIX

Table A1
New London Dump Study
Water Quality Data Cruise NL 27 5 Aug. 1976

Station	Time EDT	Depth (M)	Temp °C	Salinity ‰	Susp. Solids mg/l	Volatiles Solids % Total	O ₂ ml/l	O ₂ % Sat	TOC	TDOC
S1	1025	0		30.502	1.92	36.4	5.6	99.1		
		9		30.512	1.75	20.0	5.3	92.7		
		18		30.610	3.32	27.7	5.4	95.4		
C1-1	1056	0	18.4	30.425	1.93	35.0	5.3	93.8		
		10	18.0	30.559	2.08	20.0	5.3	92.6		
		20	17.6	30.711	4.07	12.0	5.1	89.6		
N1	1119	0	19.2	30.495	4.15	36.4	3.1	54.8		
		9	18.2	30.491	3.03	15.9	3.5	62.1		
		18	17.6	30.723	2.95	16.2	3.0	53.2		
N2	1140	0	19.1	30.521	2.91	30.9	5.1	91.7		
		7	18.4	30.560	2.88	15.5	5.4	94.9		
		15	18.0	30.628	2.67	11.9	5.2	91.6		
N3	1204	0	20.5	27.921	2.89	36.6	5.4	96.9		
		5	19.2	30.386	2.08	30.0	5.1	91.0		
		10	18.0	30.538	3.5	14.5	5.2	90.9		
W3	1230	0	18.9	30.080	2.5	22.5	5.4	95.3		
		13	18.1	30.431	2.32	20.1	5.2	90.9		
		25	18.4	30.447	2.59	14.6	5.2	91.8		
W2	1253	0	19.9	30.409	3.31	26.7	5.4	97.2		
		13	18.4	30.444	2.18	19.7	5.2	91.7		
		25	18.1	30.617	1.52	15.9	5.1	89.1		
W1	1309	0	19.5	30.469	2.46	16.9	5.2	94.4		
		11	18.3	30.496	1.97	15.0	5.1	89.1		
		19	18.0	30.637	1.93	27.9	4.9	85.6		

Table A1 (cont.)

Station	Time EDT	Depth (M)	Temp °C	Salinity ‰	Susp. Solids mg/l	Volatiles Solids % Total	O ₂ ml/l	O ₂ % Sat	TOC	TI/OC
C1-2 S M B	1325	0	19.3	30.478	2.57	29.0	5.5	98.5		
		11	18.3	30.490	2.26	19.9	5.7	100.7		
		19	18.1	30.558	3.14	16.7	5.1	89.7		
E1 S M B	1336	0	19.5	30.431	2.21	22.5	5.6	100.0		
		11	18.3	30.616	2.50	18.2	5.1	90.0		
		22	18.1	30.610	3.38	12.0	4.9	86.8		
E2 S M B	1348	0	20.0	30.475	3.22	22.9	5.4	98.7		
		10	18.2	30.546	2.19	17.9	5.0	88.7		
		20	18.0	30.729	3.32	11.1	5.0	87.1		
E3 S M B	1404	0	19.3	30.505	2.06	18.1	5.4	96.7		
		6	18.3	30.536	2.47	10.0	5.2	92.5		
		10	18.7	30.600	3.10	13.9	5.3	93.5		

Table A2

New London Dump Site
Acid Soluble Metals Sept., Oct. 1975

Station	Cd	Cu	Fe x 10 ³	Ni	Pb	Zn	Hg
A1	.2	2.5	7.1	13	6	25.0	.025
A2	<.2	13.0	8.4	6	5	47.2	N.S.
A3	.3	15.1	12.7	14	6	61.3	.003
A4	1.3	20.0	11.6	7	19	62.9	<.003
A5	<.2	16.6	11.1	10	<2	58.8	<.003
A7	.3	20.8	18.1	13	8	66.4	<.003
A8	<.2	13.8	11.4	13	24	46.5	<.003
A9	<.2	10.1	6.4	8	9	42.1	.012
A10	<.2	4.7	5.9	<1	2	33.8	.040
B1	<.2	5.5	6.1	4	<2	33.7	.032
B2	.3	5.0	5.9	3	5	31.4	.040
B3	<.2	7.9	13.5	5	10	57.7	.075
C1	<.2	10.7	10.0	10	<2	40.0	.037
C2	<.2	5.7	15.0	2	<2	47.1	.064
C3	.5	9.7	10.5	10	<2	47.5	.046
C4	<.2	11.0	8.1	<1	4	42.6	.011
C5	<.2	11.7	10.3	24	12	44.4	.017
C6	.6	14.4	29.4	26	3	42.9	<.003
C7	.6	24.3	12.3	9	30	61.3	<.003
C8	<.2	23.1	18.5	14	20	47.9	<.003
C9	.3	9.8	5.5	<1	24	42.9	.031
D2	<.2	1.1	1.7	<1	<2	9.3	.010
D3	.3	11.6	12.6	8	<2	54.9	.065
E1	<.2	2.7	2.8	6	<2	13.6	.024
E2	<.2	5.9	5.8	<1	16	19.6	.025
E3	<.2	10.7	6.3	7	<2	48.1	.039
E4	.3	14.8	4.8	8	<2	43.8	.076
E5	.3	19.2	8.9	7	<2	57.0	.008
E7	<.2	15.3	16.2	15	<2	50.3	<.003
E8	.2	10.6	10.8	9	3	39.4	<.003
F3	<.2	9.3	13.1	18	<2	47.5	<.003
F4	.2	32.9	18.8	16	62	125.2	<.003
F5	<.2	25.8	13.5	15	42	79.3	<.003
F7	.9	20.9	12.1	13	12	98.9	.031
F8	.4	20.9	7.5	7	6	42.1	.013
F9	<.2	6.4	7.7	4	9	31.9	.054

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Table A3

New London Dump Site
Acid Soluble Metals Jan., Feb. 1976

Station	Cd	Cu	Fe x 10 ³	Ni	Pb	Zn	Hg
A1	<.2	3.7	6.3	2.4	9	23.9	.020
A3	<.2	10.8	10.1	3.5	17	44.5	.056
A9	<.2	9.4	8.2	3.9	27	47.6	.038
A10	<.2	5.6	5.5	4.4	8	35.9	.050
C1	<.2	8.7	6.8	7.8	16	36.2	.034
C3	<.2	19.5	14.1	10.9	27	69.0	.015
C6	<.2	10.9	22.1	6.9	11	47.1	<.003
C7	<.2	19.6	16.4	8.4	18	60.2	<.003
C9	.3	10.9	9.0	<.8	15	48.3	.044
F3	<.2	13.8	12.4	11.0	14	49.1	<.003
F8	<.2	9.1	8.4	9.9	12	34.9	.018

E. Effect of Dredging and Spoils Deposition
on Fecal Coliform Counts in Sediments
and Bottom Waters of the Thames River
and New London Disposal Site

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Introduction

Fecal coliforms, indicators of fecal pollution and enteric pathogens, have been shown to persist and concentrate in bottom sediments in aquatic systems which have received fecal material (Rittenberg, Mittwer and Ivler, 1958, Hendricks, 1971, and Van Donsel and Geldreich, 1971). The dredging and transport of these sediments could lead to bacteriological contamination of the spoils disposal and adjacent areas.

In a hydraulic dredging operation of a navigation channel on the upper Mississippi River (Wisconsin), it was shown that the Fecal coliform (FC) concentrations increased significantly in the water column in the immediate vicinity of the operation (Grimes, 1975). Concentrations of FCs reaching the deposition site, however, were greatly diluted due to the tremendous volumes of water necessary to pump the spoils to the spoiling site.

The objective of this study was to monitor the FC densities in the Thames River sediments and at the spoils deposition and adjacent areas in Long Island Sound for evidence of fecal contamination due to the deposition of the dredged material. Also in the initial survey a comparison was made of the total aerobic

plate count and fecal coliform count of the top sediments from the stations sampled.

MATERIALS AND METHODS

Sampling area. Sediment samples were collected from five stations in the Thames River (Fig. 1) and from 30 stations (17 spoils and 13 control stations) at the New London dump site in Long Island Sound (Fig. 2). The spoils ground was considered to be those sites encompassed by a circle with a one-mile radius measured from the point of dumping (Station C6). The control stations were located outside this circumference. All stations were sampled for sediments in July, 1974, and July, 1975. Two spoils stations (C6, C4), 4 control stations (C3, E3, A3, C2) and one river station (R4) were sampled quarterly for bottom water at high and low tides and for top sediments during the 15-month period of study.

Sampling technique. Bottom sediments were collected with a Smith-McIntyre dredge. The top centimeter of the sediment surface was removed with a sterile tongue depressor and placed in a sterile 8 oz. French square bottle. Bottom water samples (one meter above the bottom) were collected using sterile bag samplers (General Oceanics Model #1030). The water samples were then transferred to sterile 64 oz. French square bottles. All sediment and water samples were immediately refrigerated and analyzed within 24 h and 12 h, respectively. The sediment temperatures were measured with a YSI Model 42SC Telethermometer.

Fecal coliform enumeration. FC densities in water were determined using the five-tube most-probable number (MPN) procedure recommended for seawater analysis (1). MPN procedures for the enumeration of FC in sediments have been described in a previous paper (2). Lauryl sulfate tryptose broth was used in the presumptive test with confirmation in EC broth incubated at 44.5 C in a circulating water bath.

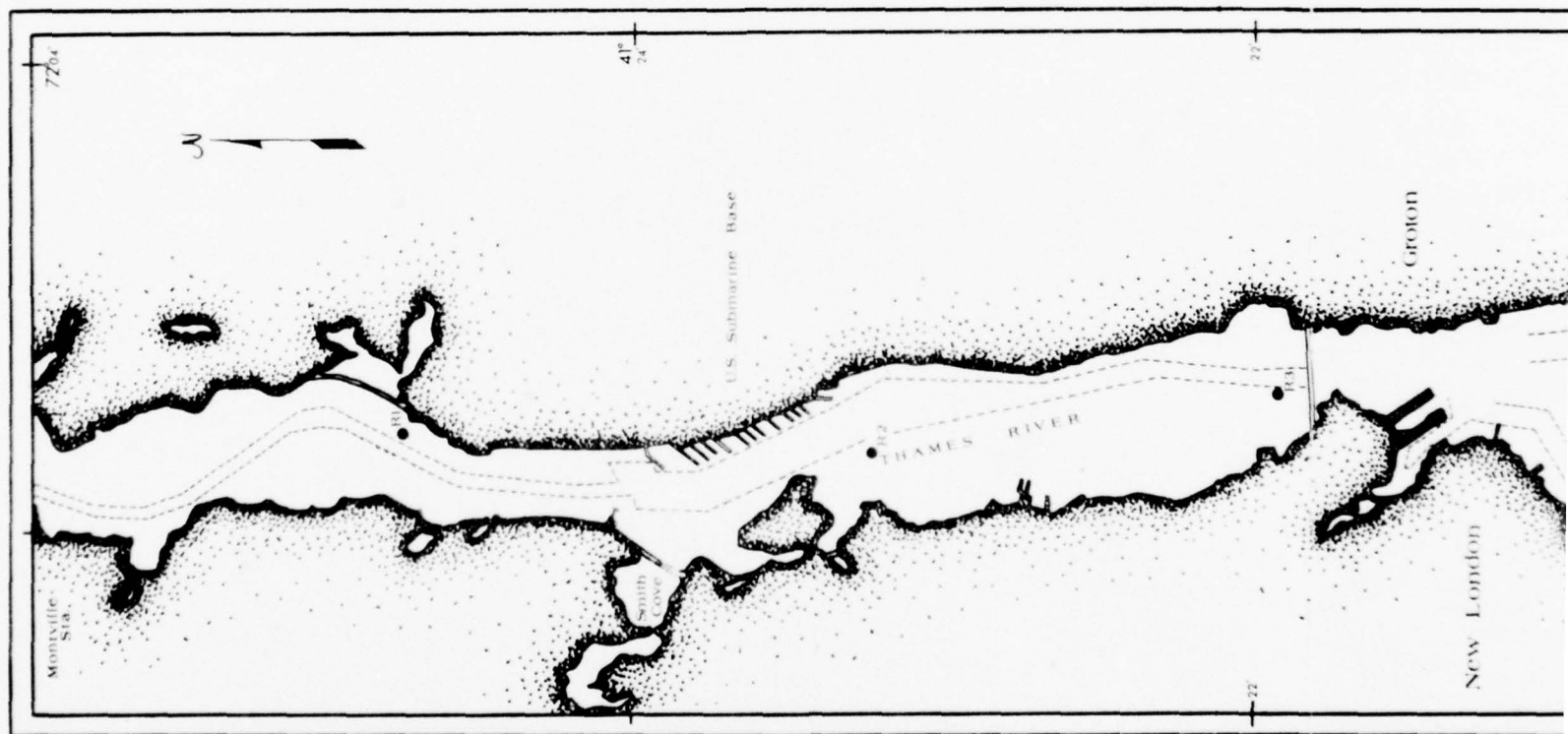
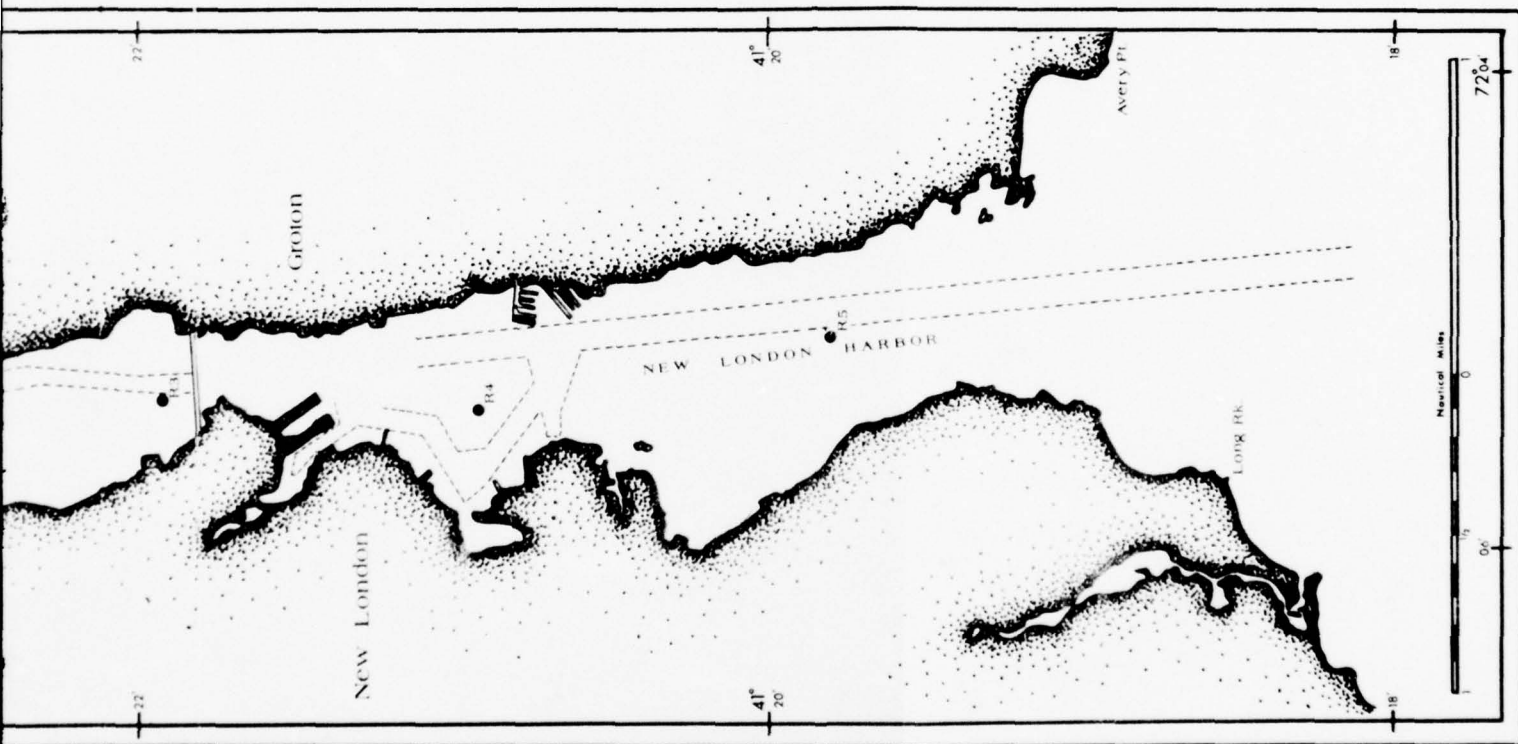


Fig. 1. Thames River locations for sampling fecal coliform bacteria in sediments.



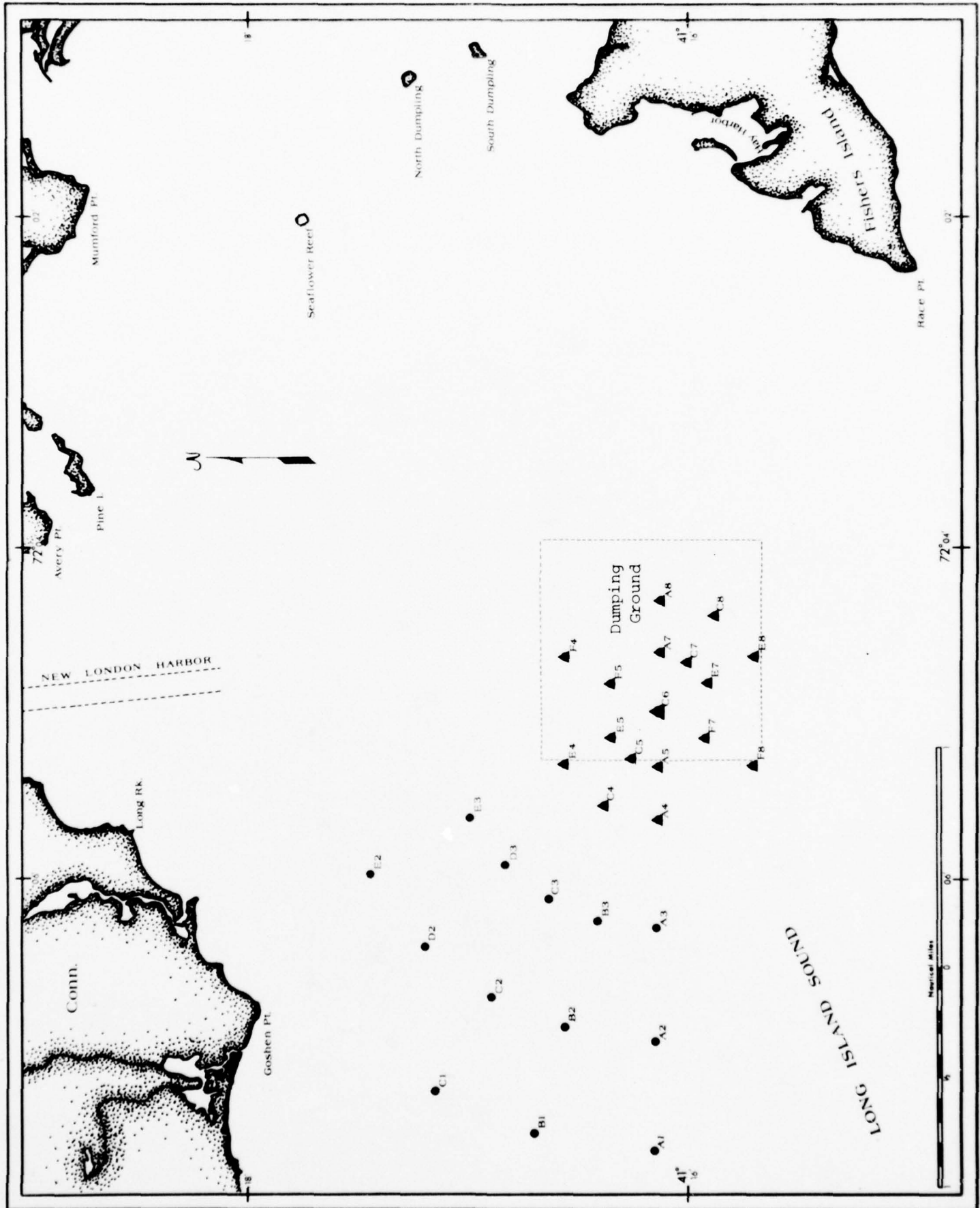


Fig. 2. Disposal area locations sampling fecal coliform bacteria in sediments. Triangles indicate "spoils" stations; circles are controls.

All dilutions used 0.1% peptone broth as the diluent.

Total aerobic plate counts were performed by serially diluting known volumes of the top sediment in cold sterile peptone water and surface plating onto a minimal nutrient sea water medium with incubation at 20°C until maximum counts were observed, usually up to 2 weeks.

RESULTS AND DISCUSSION

During the initial FC survey, prior to dredging, we determined that it was extremely important for synopticity to sample all stations within a relatively short period. As shown in Table 1, there was a great difference in FC densities between stations in transects A and C (sampled June 26 and 27) and those which were in close proximity in transect B (sampled July 8). To determine if the 12-day interval between the sampling dates was a factor for the difference found in FC concentrations between these proximate stations, all of these stations were re-sampled on July 30 and the FC analyses repeated. The data obtained from resampling the stations (Table 1) showed a uniform FC distribution pattern.

Previous work (2, 6) has shown that the stability of FC counts is dependent upon the in situ sediment temperature. Our studies have shown that the FC counts are stable for two to four days at 20 C and up to two weeks at 4 C. In this study, the mean sediment temperature at 80 stations sampled during the initial period was 15.7 ± 0.65 C. Therefore, at this temperature, there was a sufficient time interval between the sampling of the transects to account for the differences found in the FC concentrations. In any regard, in the initial survey, all stations along a transect were sampled within one day so that the comparative relationships between FC concentrations at spoils and control stations along a transect would not be affected. The subsequent quarterly samplings were made in one day, and the July, 1975, samplings were completed in 3 days.

TABLE 1. Comparison of fecal coliforms in
sediments from selected stations before dredging
operation began

Station	Fecal coliforms/100 ml	
	June, July 74 ^a	July 30, 74
A2	1,720	330
A3	11	5
A4	490	33
A5	240	330
B2	14	490
B3	2	170
B4	11	49
B5	14	70
C2	700	490
C3	4,900	240
C4	2,200	70
C5	3,300	49
C6	220	79
C7	330	130
C8	490	23
E7	26	0
E8	109	172
F7	26	17
R4	22,100	17,200

^a Sampling dates: A transect, June 26; B transect
and Station R4, July 8; C transect, June 27;
E7, E8 and F7, July 2.

Quarterly distribution of FCs in sediments during the 15-month sampling period (Table 2) demonstrated no significant differences between FC counts in the spoils and control areas. The FC concentrations in bottom sediments and water actually decreased throughout the sampling area during this period (Tables 2, 3). During the first two quarters when FC densities were highest in the bottom waters, the highest counts were found during low tide when river water inflow into the disposal area was the greatest. This factor, along with the high FC levels in river water (Table 3), indicates that the outflow of water from the Thames River plays a major role in determining the FC concentrations present in the sediment at the disposal site.

The data collected from the comprehensive sampling accomplished before the dredging operation began and one year later after dredging was completed (Table 4), showed similar geometric means for FC concentrations at the spoils and control area and the same decrease in FC counts as previously described. In addition, the FC densities in the spoils area decreased more during this period than they did in the control area. The FC counts in the river sediments also decreased. The decrease in the FCs found in the bottom waters of the river and the disposal site (Table 3) were probably responsible for the decline of the FCs found in the top sediments.

Our data indicate that the deposition of material dredged from the Thames River did not increase the FC densities in the sediments of the New London dump site. Although the FC concentrations in the Thames River sediments prior to dredging were high, these bacteria are present only in the upper layer of the sediment and are, therefore, greatly diluted by the underlying sediments during the dredging and dumping operations.

Table 5 presents the data comparing the total aerobic plate count and the

TABLE 2. Quarterly distribution of fecal coliforms in sediments

Station	Area	Fecal coliforms/100 ml						
		July 74 ^a		Oct. 74	Jan. 75	April 75	July 75	Oct. 75
C6	Spoils	220	-- ^b	49	790	310	5	33
C4	Spoils	2,200	70	330	221	460	33	79
C3	Control	4,900	240	220	130	330	33	33
E3	Control	2,210	--	490	330	330	49	70
A3	Control	11	5	170	33	490	14	49
C2	Control	700	490	3,300	4,900	490	79	79
R4	River	22,100	17,200	79,000	4,900	70,000	240	27,000

^a Selected stations sampled twice.

^b --, No data.

TABLE 3. Distribution of fecal coliforms in bottom waters at low and high tides

Station	Area	Fecal coliforms/100 ml							
		July 74		Oct. 74		May 75		July 75	
		Low	High	Low	High	Low	High	Low	High
A3	Control	8	2	11	2	--	--	--	--
E3	Control	79	0	8	--	0	0	--	--
C2	Control	-- ^a	--	8	--	2	0	8	0
C3	Control	49	5	13	2	2	0	5	0
C4	Spoils	--	--	5	2	0	0	0	0
C6	Spoils	2	0	8	2	0	0	0	5
R4	River	630	790	490	172	--	130	49	130

^a --, No data.

TABLE 4. Geometric means of fecal coliform counts in sediments before and after phase I dredging operation

Sampling area	Number of stations sampled	Geometric mean Fecal coliforms/100 ml	
		July 74 ^a	July 75
Spoils	17	170	6
Controls	13	220	40
River	5	14,000	810

^a Includes FC counts of stations sampled twice, six in control area and nine in spoils area.

Table 5 Distribution of total bacterial counts and fecal coliforms in sediments from Thames River-New London Dumpsite- June 26 - July 8, 1974.

Station	<u>Initial Survey</u>	
	Total Plate Count X 10 ⁴	Fecal Coliforms /100 ml
A1	69	1,300
A2	110	1,720
A3	140	11
A4	140	490
A5	70	240
A6	100	221
A7	72	790
A8	85	460
A9	160	1,410
B1	33	17
B2	26	14
B3	36	2
B4	26	11
B5	15	14
C1	130	172
C2	170	700
C3	160	4,900
C4	150	2,200
C5	95	3,300
C6	17	220
C7	95	330
C8	120	490
C9	93	490
D2	37	221
D3	72	172
D4	91	221
D5	37	130
E1	970	7,900
E2	260	2,400
E3	290	2,210
E4	160	1,090
E5	250	490
E7	310	26
E8	150	109
E9	130	22
F3	140	790
F4	200	460
F5	890	490
F7	170	26
F8	110	172
F9	37	172
R1	2,650	1,300
R2	2,200	4,900
R3	1,080	172,000
R4	6,500	22,100
R5	930	24,000

FC count in the top sediments obtained from the stations sampled during the initial survey. Analyses of total aerobic plate counts showed no set pattern of distribution. The bacterial densities were significantly higher in river sediments than from those obtained from the dumpsite and adjacent area. In general the highest plate counts of the sediments were observed from the Northern section relative to the dumpsite, indicating contributions of the river outflow.

Comparison of total counts with the fecal coliform counts in sediments from each station did not yield a high degree of correlation.

SUMMARY

Fecal coliform concentrations in surface sediments were monitored at the New London dump site in Long Island Sound during the deposition of the dredge spoils from the Thames River. Although the geometric mean for fecal coliform at five stations in the river was 14,000/100 ml before dredging commenced, the deposition of this material did not increase the incidence of fecal coliforms at 17 experimental stations and 13 control stations located in the spoil disposal and surrounding areas. Fecal coliform analyses of bottom waters during flooding and ebbing indicated that the outflow of water from the Thames River played a major role in determining the concentration of fecal coliforms found in the sediments at the disposal and control sites. Comparisons of the total aerobic count with the fecal coliform counts with sediments did not yield a high degree of correlation.

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F. Sediments and Benthic Macrofauna
of River and Disposal Area

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INTRODUCTION

Benthic macrofauna, due to their relative immobility and wide range of life histories, have commonly been held to be a most appropriate group of organisms for use in biological monitoring of impacts (Wilhm, 1967; Reish, 1972; Boesch, 1974). Knowledge of impacts on the benthic macrofauna, in turn, permits estimation of effects on food webs used by resource species, and of contaminant flow through those food webs. A number of studies have examined impacts of dredging and spoiling on the benthos; a comprehensive review is given by Morton (1976).

The background and objectives of the present study were discussed in the general Introduction to this report. The sediments and benthic macrofauna project concentrated on analysis of macrofauna samples taken quarterly in disposal and control areas, in an attempt to determine the spatial and temporal extent of any disposal effects. Macrofauna of the Thames River were studied to examine dredging impacts, as well as to determine whether presence of riverine fauna in the disposal area could be used as an indicator of spoil movement. Sediments from both river and

disposal area were analyzed to trace spoil migration and to better understand macrofaunal distributions.

Some information on the area's sediments and benthic macrofauna was available prior to the present survey. Assessments of the Thames River's sediments and biota had been made to predict impacts of the dredging, and are presented in the New London Final EIS and its supplement (Department of the Navy, 1973, 1976). An intensive environmental survey of the river in the vicinity of the U. S. Coast Guard Academy has also been completed (Tolderlund, 1975). In the disposal area, the Northeast Fisheries Center (NEFC), Sandy Hook Laboratory, has studied sediments and benthic macrofauna of a station 0.4 nautical miles (n mi), (0.74 km) south of the disposal buoy, and of another site near the Thames' mouth, since 1972 as part of a baseline survey of the benthic ecology of Long Island Sound (Reid et al., subm. for publ.). The University of Connecticut (UCONN), Avery Point, Groton, Conn., has undertaken a series of studies of the benthic macrofauna of nearby Fishers Island Sound (Franz, 1976). In a study pertinent to the present work, the Naval Oceanographic Office (1973) conducted surveys before, during and after a "test" dump of $92,500 \text{ yd}^3$ ($70,855 \text{ m}^3$) of spoils from the Thames at the New London Dumping Ground. We have used findings of these surveys to help establish "baseline" conditions for assessment of effects of the much larger project presently under study.

Concurrently with our monitoring survey, the Naval Underwater Systems Center (NUSC) had been investigating currents and bathymetries of the disposal area (Morton, Cook and Massey, 1975). Pertinent results of the NUSC studies are discussed below.

METHODS

The disposal area sampling pattern consisted of six transects of stations radiating out from the disposal buoy (Figure 1). Station C6 was located at the buoy, and others were established at distances 0.25, 0.5, 1 and 2 n mi from the buoy. Station density was highest to the north and west of the disposal point; this reflects the fact that the literature available at the study's inception described net bottom currents in eastern Long Island Sound as flowing generally to the northwest (Riley, 1952, 1956; Gross and Bumpus, 1972). More recent studies concerned specifically with the New London dumping grounds (Morton, Cook and Massey, 1975; Hollman, Section C of this report) have described net bottom movements to the south and east. Station A10 was thus added after the first sampling period to monitor impacts in the vicinity of Fishers Island. Six stations in the Thames River (R1-R6) spaced at 1-1.5 n mi intervals from the river mouth to 0.7 n mi above the northern limit of the submarine base, were also occupied (Figure 2). These stations were located immediately outside the intended dredging areas, so that impacts other than the gross removal of faunal assemblages with the spoil could be examined. A seventh "river" station (R-7) was established 0.4 n mi east of the seaward end of the river channel. In all, a total of 46 stations were sampled.

Bottom waters, sediments and benthic macrofauna of all stations were sampled at least six times, at approximately quarterly (3 month) intervals. The baseline sampling was conducted from 26 June to 12 July 1974. Dates for subsequent samplings were: 23 September to 4 October 1974;

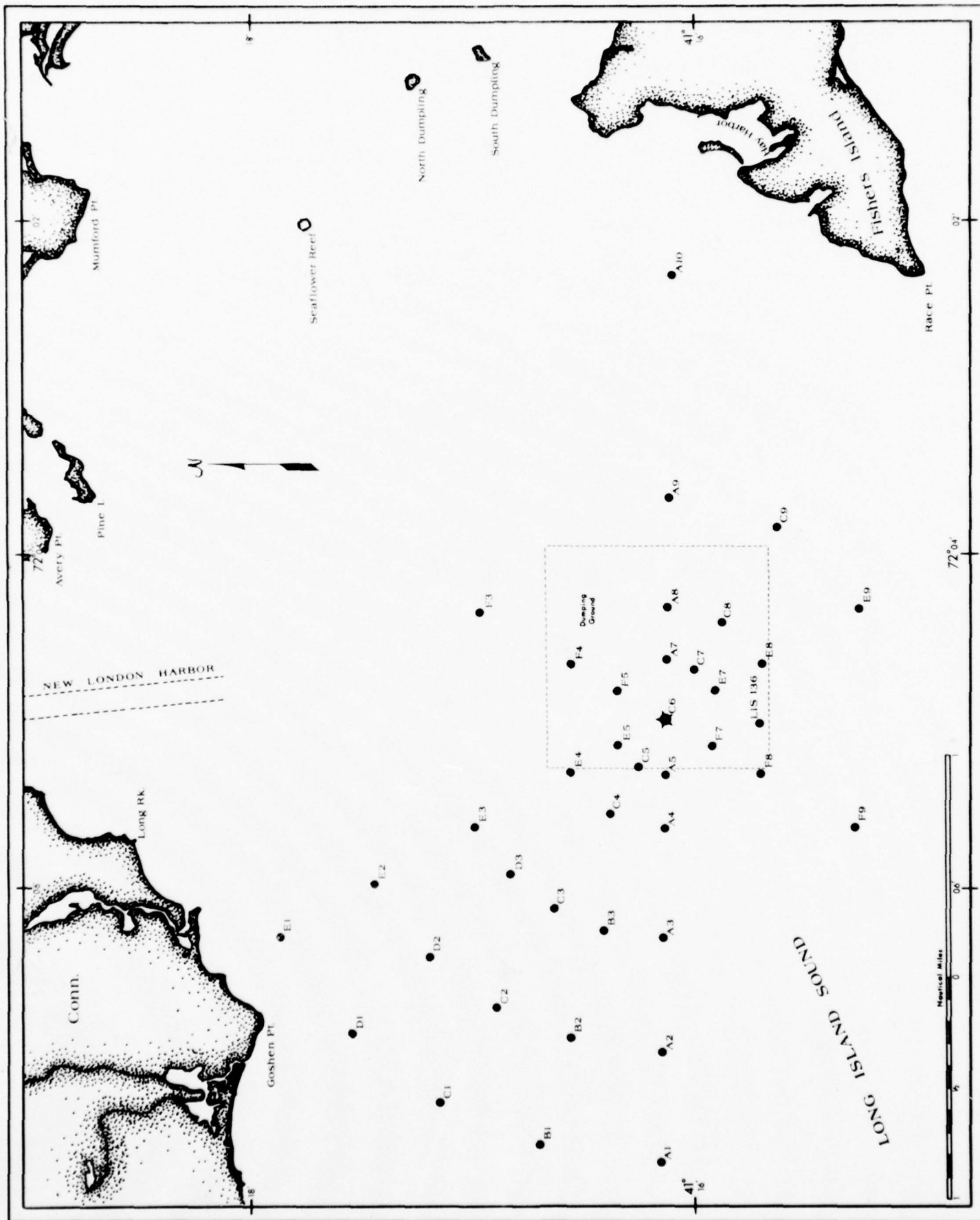


Figure 1. Disposal area sampling pattern for sediments and benthic macrofauna. ★ = approximate spoil disposal point.

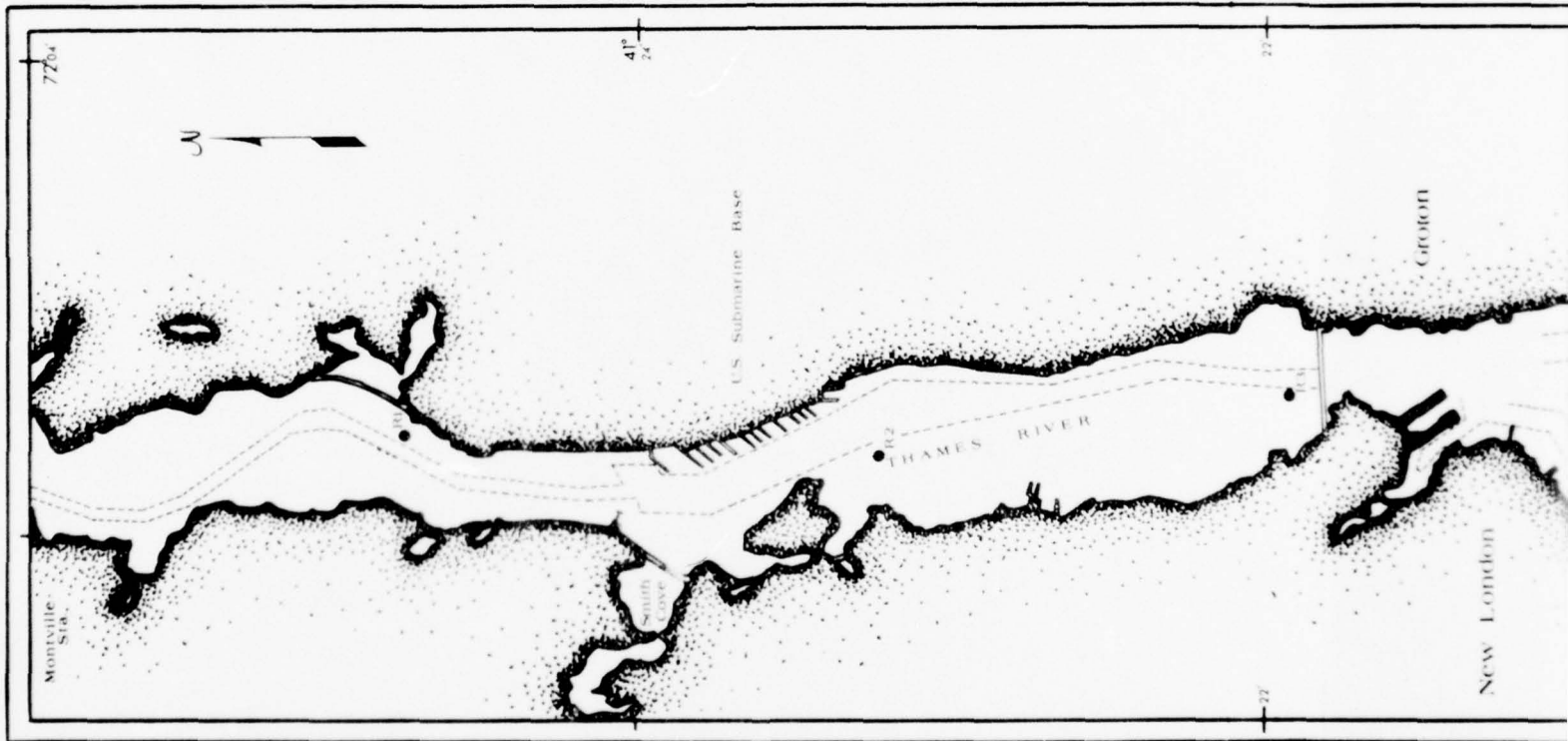
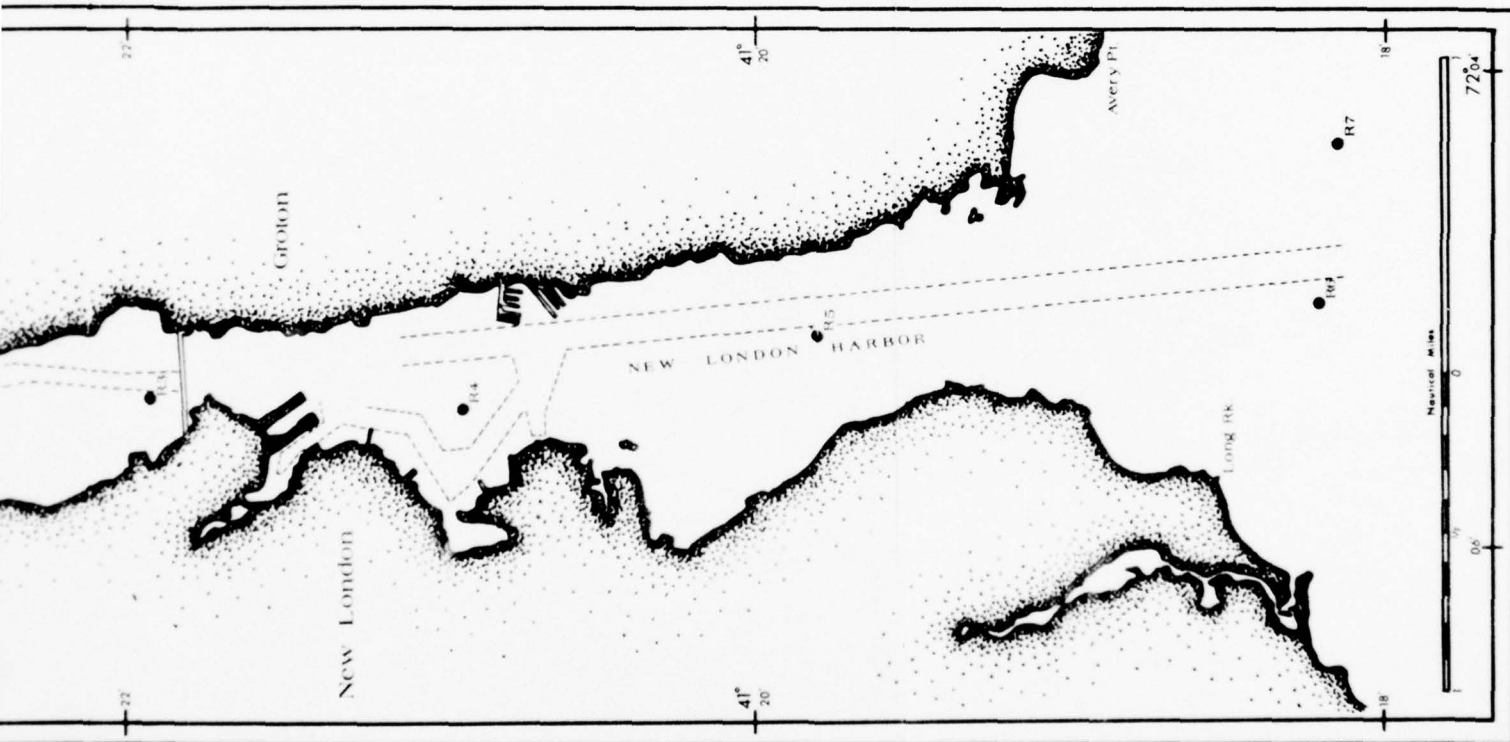


Figure 2. Thames River sampling pattern for sediments and benthic macrofauna.



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20 January to 4 February, 21 April to 1 May, 23 June to 30 July and 23 September to 1 October, 1975. As a rule, SCUBA surveys to observe sediments and megafauna, and to measure sedimentation rates, followed shortly after the remote samplings. Further reference to these sampling periods will be by the month in which a sampling began, i.e., June 1974.

Station location was achieved by combined use of radar and fathometer, augmented by horizontal sextant and by land and buoy ranges when possible. Niskin bottles were used to collect water samples from 1 m above bottom at each station for analysis of bottom salinity and dissolved oxygen levels. Salinities were determined using a Beckman RS-7C Induction Salinometer.* The azide modification of the Winkler technique (American Public Health Association, 1965), with PAO titrant (Hach Chemical Co., Ames, Iowa), was used for dissolved oxygen determinations.

A Smith-McIntyre bottom grab, sampling 0.1 m^2 of sediment surface, was used to collect five replicate samples per station during each sampling period. Bottom temperatures were measured by inserting a thermometer into the sediment of the grab samples. A subsample was collected from each grab with a plastic coring tube (3.7 cm inner diameter, sampling to a depth of ca. 15 cm) and frozen for sediment analyses.

The sedimentology (grain size distribution, % organics, % carbonates) was performed under contract by Dr. James Parks and Mr. Alex Rugh, Lehigh

*Use of trade name does not imply endorsement by the National Marine Fisheries Service.

University, Bethlehem, Pa. For grain size analysis, a portion of each core sample was wet-sieved through a 62 μ screen, with retained materials then sieved through a series of 12 screens ranging from 4 mm to 62 μ mesh. Pipette analysis was used to determine the coarse silt, fine silt and clay fractions of the $\leq 62 \mu$ portion. Calcium carbonate and organic contents were determined by weight loss of dried samples after treatment with dilute hydrochloric acid and 10% hydrogen peroxide, respectively.

The New York Ocean Science Laboratory (NYOSL) Montauk, Long Island, N. Y., also obtained materials from the grab samples for analysis of sediment constituents and of heavy metals in macrofauna (Section D). The remainder of the sample was rinsed through a 1 mm mesh sieve, with the residue preserved in a 1:9 formalin to seawater mixture and later transferred to 70% ethanol with 5% glycerin.

Samples were sorted under dissecting microscopes. Organisms were identified to species when possible. For identification and nomenclature, we consulted primarily Pettibone (1957, 1963) for polychaetes; Abbott (1954, 1968) for molluscs; Schultz (1969) for isopods; McCain (1968) and Bousfield (1965, 1973) for amphipods; Williams (1965) for decapods and Gosner (1971) for other taxa. Confirmation of identifications was by one of the authors (ABF). Certain groups have been omitted from the macrofaunal analysis due to their infrequent occurrence, predominantly planktonic nature, uncertain identification and/or difficulty in quantification. We have concentrated on four invertebrate classes, Polychaeta, Gastropoda,

Bivalvia and Crustacea. These classes together comprise an estimated 95% of species, and also a great majority of individuals, of macro-fauna (<1 mm) organisms we have collected in the New London area. We also have included pycnogonids, tunicates, echinoderms, sipunculids and a single anemone, Metridium senile, in our analysis, since these forms were numerous enough to be of possible ecological importance at one or more stations.

It has not been possible to process completely all the benthic material generated by this intensive sampling program. We have concentrated our efforts on selected samples from among three groups of disposal area stations: 1) Those in the spoil pile or its immediate proximity; 2) Stations at 1 n mi radii from the disposal buoy; as identified in the monitoring criteria, this is a critical distance in terms of detecting impacts; and 3) "Control" stations 2 n mi from the disposal buoy on selected transects. Key stations from these three categories (A1, A3, A9, C1, C3, C6, C9, F3 and F8) have been sampled a seventh and eighth time, in February and in September 1976. As a minimum, these stations will be resampled yearly to assess longer-term impacts and recolonization.

Species diversities were calculated using the Shannon and Weaver (1963) index: $H' = -\sum_{i=1}^S p_i \ln p_i$, where p_i is the proportion of individuals in the i^{th} species. H' has two components: number of species (S , hereafter termed species richness), and equitability ($J' = H'/H'_{max} = H'/\ln S$) (Pielou, 1975). Equitability is a measure of the evenness of

distribution of individuals among species. Both S and J' were computed for each 0.1 m^2 sample processed, as was number of individuals (N).

To examine effects of spoiling on species composition, Q - mode cluster analysis (stations compared based on similarity of organisms found there - Orloci, 1967) was performed on all June 1974 samples, and results compared to similar analyses of June 1975 data. Czekanowski's coefficient, $C_z = 2w/a+b$ (Bray and Curtis, 1957) was used to measure faunal similarity between stations. Here "a" is the sum of abundances of all species found at station A, "b" is the sum of species abundances for station B, and "w" is the sum of the lower of the two abundance values for each species common to A and B. Abundances were transformed by natural logarithms and then clustered using unweighted pair grouping (group average sorting - Sokal and Sneath, 1963).

When replicate grabs were processed, mean values for species abundances were used in the cluster analysis. There are problems inherent in this approach; for instance, cumulative S in the pooled grabs was always considerably greater than S of any individual grab. This could invalidate direct comparison with single-grab stations. To make the data as comparable as possible, for each replicated station we eliminated rare species from the analysis of that cutoff density at which cumulative S remaining in the analysis most closely approached the mean S for that station (i.e., at A1 in June 1974, cumulative S for five replicate samples was 42, and mean S , 23.4. By omitting all species with a mean density of less than 1.2 individuals per 0.1 m^2 , cumulative S for the cluster analysis was reduced to 23). The cutoff densities for the replicated stations ranged

from 0.6 to 1.4 individuals per 0.1 m^2 , and were in general quite close to the 1.0 minimum density of unreplicated samples. We did not follow the procedure, sometimes used in cluster analysis, of eliminating species occurring at less than a specified percentage of stations. Analyses were run on an IBM 360-65 computer, using a program obtained from Dr. Donald Boesch of the Virginia Institute of Marine Sciences, Gloucester Point, Va., and modified by Dr. Donald Maurer, College of Marine Sciences, University of Delaware, Lewes, Del.

RESULTS AND DISCUSSION

Sediments

The typical sediment type at disposal area stations prior to the present spoiling was silty sand ($\bar{X} = 36.5\%$ silt-clay, or materials $< 62.5 \mu$), with moderate organic material (range 0.1-8.9%) and variable carbonate content (1.3-27.9%). Stations differing most markedly from this description were A1 and C1, where sediments were almost entirely medium and fine sands, and F4, with 78.6% silt-clay. The disposal buoy station, C6, had a representative predisposal composition. Table 1 presents the grain size profile, organic and carbonate contents of surface sediments at C6 as they varied during the course of our study. Effect of spoiling is shown by the large differences in grain sizes between June (predisposal) and September 1974 after dumping had already begun; all subsequent samples have shown size distributions very similar

Table 1. CHARACTERISTICS OF SURFACE SEDIMENTS AT ORIGINAL NEW LONDON DISPOSAL POINT

DATE	DIAMETER, MM MEAN MEDIAN	SORTING INDEX	GRANULE 4.0mm	WEIGHT PERCENT RETAINED ON MESH SIZE IN MM										Wt. % ORGANICS	Wt. % CARBONATE
				SAND			SILT			CLAY					
				2.0	1.0	0.5	0.25	0.125	0.0625	0.0156	0.004	<0.004mm			
26 Jun 74	.055	.078	-	1.75	1.65	3.65	9.47	18.29	19.57	23.40	13.68	8.53	1.11	3.19	
26 Sep 74	.016	.019	-	-	-	.15	.25	.27	1.04	56.28	29.59	12.42	3.17	2.57	
24 Jan 75	.018	.022	-	-	.33	.58	.56	.73	3.13	59.41	23.87	11.39		2.75	
23 Apr 75	.016	.020	-	.14	.03	.41	.70	.70	3.72	54.60	24.78	14.91	1.12 ^a	6.38 ^a	
24 Jun 75	.017	.021	-	-	-	.18	.30	.56	3.72	57.75	24.70	12.78	5.29	4.48 ^a	
23 Sep 75	.016	.020	-	.15	.03	.19	.56	.69	1.17	55.60	28.22	13.41		6.19	

^amean of two samples

to the September 1974 values. The most obvious change is the increase in silts and clays with spoiling. All September 1974 and subsequent samples from C6 contained at least 94% silt-clay, a value substantially larger than that of the predisposal sediments from any station.

Other selected stations throughout the disposal area were, therefore, analyzed for change in percent silt-clay in an attempt to detect presence and/or spread of spoils. As Table 2 shows, temporal variability in percent silt-clay was high at many stations. This variability is partly an artifact of imperfect station relocation, but it also demonstrates the real "patchiness", or small-scale spatial heterogeneity, of disposal area sediments. Large differences were found even between replicate samples taken while at anchor or while returning to an anchored buoy on a single tide. Despite this variability, it can be concluded from Table 2 that only station C6, and perhaps A5, C7 and C8, showed consistent, long-term increases in % silt-clay indicative of buildup of spoils. Also, if at any time the bottom grab had encountered a layer of spoils thicker than its ca. 15 cm sampling depth, the sample's silt-clay content would presumably resemble the $\geq 94\%$ values at C6. This was not found for any disposal area sample taken outside of C6. As expected, silt-clay content of Thames River stations R1 - R5 (Table 2) is comparable to that found in the spoil pile.

Macrofauna Community Structure

A total of 340 grab samples were analyzed. Values for N, S, H'

Table 2. Temporal changes in percent silt-clay (<0.0625 mm) of surface sediments at selected stations in disposal area and in Thames River ("R" stations).

STATION	JUN 1974	SEP 1974	JAN 1975	APR 1975	JUN 1975	SEP 1975
A1	8.5	12.9	11.6	10.9 ^a	5.7 ^a	4.5 ^a
A3	42.7	15.4	30.3	51.9 ^a	66.6 ^a	38.2 ^a
A4		55.3 ^a	47.1	62.6 ^a	59.0 ^b	42.1 ^a
A5	32.4	50.1	41.9	48.1 ^a	89.0	51.0 ^a
A8	25.0	90.6	91.8	52.1 ^a	40.8 ^a	26.5 ^a
A9	37.4	21.1	38.0	17.2 ^a	27.6 ^a	39.4 ^a
B3	42.3 ^a	27.1 ^a		63.7 ^a	32.4	31.2 ^b
C1	7.7	55.1	17.1	26.5 ^a	35.3 ^a	47.9
C2					19.8	
C3	41.9	22.2	24.7	23.5 ^a	24.2	16.7 ^b
C4	32.2	18.4	32.7	38.1 ^a	55.2 ^a	25.4 ^a
C5	67.1	56.0	43.6	83.8 ^a	43.0 ^a	29.4 ^a
C6	45.6	98.3	94.7	94.5 ^a	95.3 ^a	97.2 ^a
C7	22.0	58.2	62.5	84.8 ^a	78.5 ^a	52.4 ^a
C8		22.1 ^a		66.0 ^a	79.3 ^a	72.9 ^a
C9	34.6	49.5	27.1	48.0 ^a	61.1 ^a	25.7 ^a
E3		20.4	16.2	18.4 ^a	15.5	21.2 ^a
E8			39.4			
E9		14.8	17.0	11.2 ^a	15.1	22.3 ^a
F3	38.6	22.0	24.8	30.3 ^a	17.6	14.4 ^a
F4	78.6	37.4	57.7	76.1 ^a	24.5	32.1 ^a
F8	27.6	39.1	43.7	90.1 ^a	44.5 ^a	22.6 ^a
R1	97.8					
R2	92.8					
R3	97.5					
R4	94.5					
R5	93.1					
R6				77.7		
R7	32.7					

^amean of two samples; ^bmean of three samples; other values represent single core samples.

and J' for all stations and dates processed are presented in Table 3. Where replicate samples were processed, values reported are means. As the June 1974 values indicate, the New London area supported a fairly rich and diverse fauna at the time of the baseline survey. The overall mean N of 470.3 individuals/ 0.1 m^2 was comparable to densities reported for Block Island Sound (Steimle, et al. 1976) and Rhode Island Sound (Saila, Pratt and Polgar, 1972) though it was somewhat lower than densities found in the soft-bottom areas of Long Island Sound in 1972 (Reid, et al., subm. for publ.) Mean S (41.7) and H' (2.44) were higher than those of Rhode Island Sound or central Long Island Sound. Diversities were similar to values given for shallow shelf areas off Virginia in Boesch's (1972) review of diversity patterns. The relatively high diversities may not bear a simple relationship to environmental stability or "health", however (Goodman, 1975). J' ranged widely, from .259 to .878; the lower values corresponded to dense populations of mussels (at C1) or amphipods (several stations). Species composition is discussed in more detail below.

For a statistical examination of impacts of spoiling on community structure, we calculated ninety-five percent confidence limits or CLs (taken as the product of standard error and the appropriate value from a two tail t table) for stations with three or more grabs processed per sampling (Table 3). Since there were large, apparently natural, seasonal changes in macrofauna populations, we limited comparisons of the data to the same months in succeeding years. Brackets are used

Table 3. Mean values for number of individuals (N), number of species (S), species diversity (H'), and equitability (J'), from 0.1 m² Smith-McIntyre bottom grab samples. 95% confidence limits given in parentheses; bracketed values denote areas in which confidence limits for comparable periods of succeeding years do not overlap. n = number of replicate samples used in calculations.

Station	Date	N	S	H'	J'	n
A1	June 74	116.8 (46.9-186.7)	23.4 (13.8-33.0)	2.23 (1.34-3.11)	.708 (.451- .966)	5
	Sep. 74	[24.7 (- 1.9- 51.2)]	12.0 (2.1-21.9)	2.16 (1.44-2.89)	.887 (.730-1.04)	3
	Jan. 75	75	19	2.47	.838	1
	Apr. 75	30	13	2.26	.879	1
	Jun. 75	38.2 (4.8- 71.6)	16.8 (6.2-27.4)	2.42 (1.85-2.99)	.901 (.843- .959)	5
	Sep. 75	[109.5 (78.8-140.2)]	29.5 (19.2-39.8)	2.79 (2.54-3.04)	.828 (.838- .819)	4
A3	Jun. 74	614.0 (232.4-995.7)	50.5 (35.1-65.9)	2.74 (1.90-3.57)	.697 (.530- .864)	4
	Sep. 74	175.8 (103.6-247.9)	27.5 (18.3-36.7)	2.26 (1.95-2.57)	[.686 (.626- .746)]	4
	Jan. 75	207.2 (- 2.0-416.4)	28.2 (17.1-39.3)	[1.79 (1.36-2.22)]	[.546 (.424- .667)]	5
	Apr. 75	102.2 (26.6-177.8)	28.0 (18.4-37.6)	2.59 (2.44-2.75)	.791 (.708- .874)	5
	Jun. 75	152.3 (- 9.8-314.3)	40.0 (15.9-64.1)	3.12 (2.46-3.78)	.865 (.761- .969)	4
	Sep. 75	134.0 (78.6-189.4)	32.2 (21.1-43.3)	2.78 (2.49-3.08)	[.810 (.759- .861)]	5
	Feb. 76	129.3 (111.3-147.3)	30.3 (18.1-42.6)	[2.84 (2.38-3.30)]	[.834 (.795- .873)]	3
	Apr. 75	124	29	2.38	.706	1
A7	Sep. 75	107	16	1.77	.638	1
A8	Apr. 75	285	23	1.13	.361	1
A9	Jun. 74	458.3 (178.5-738.0)	39.5 (31.2-47.8)	2.01 (1.37-2.66)	.529 (.407- .651)	4
	Sep. 74	442.8 (323.1-562.4)	33.3 (28.7-37.8)	1.68 (1.47-1.89)	.479 (.422- .535)	4
	Jan. 75	381.6 (285.0-478.2)	33.8 (27.9-39.7)	1.65 (1.29-2.01)	.469 (.384- .555)	5
	Apr. 75	406.0 (269.7-547.3)	35.2 (31.3-39.1)	1.66 (1.41-1.91)	.466 (.398- .533)	5
	Jun. 75	344.4 (155.9-533.0)	34.6 (20.1-49.1)	2.18 (1.81-2.55)	.621 (.566- .667)	5
	Sep. 75	584.0 (447.0-720.9)	38.0 (28.1-47.8)	1.57 (1.30-1.85)	.434 (.368- .500)	5
	Feb. 76	667.7 (367.9-967.5)	33.7 (24.3-43.1)	1.25 (0.93-1.58)	.357 (.258- .456)	3

Table 3. (continued)

Station	Date	N	S	H'	J'	n		
A10	Sep. 74	137.7	(662.7-2019.3)	26.7	(17.3- 36.1)	1.13 (0.86-1.41)	.345 (.275- .415)	3
	Apr. 75	397		20		0.95	.317	1
C1	Jun. 74	1471.4	(294.8-2648.0)	40	(33.9- 46.2)	0.94 (0.42-1.47)	.259 (.104- .415)	5
	Sep. 74	946.7	(753.8-1139.5)	36.7	(22.1- 51.2)	[0.82 (0.42-1.21)]	[.228 (.104- .352)]	3
	Jan. 75	60.5		15.5		1.70	.669	2
	Apr. 75	288		25		0.93	.289	1
	Jun. 75	382.3	(-164.9- 929.5)	30.7	(26.9- 34.5)	1.82(-0.03-3.67)	.533(-.031-1.107)	3
	Sep. 75	482.3	(61.9- 902.6)	37.3	(21.6- 52.9)	[2.19 (1.47-2.90)]	[.605 (.475- .735)]	4
C3	Jun. 74	[169.4	(73.8- 265.0)]	38.0	(23.3- 52.7)	3.00 (2.34-3.65)	.831 (.697- .965)	5
	Sep. 74	503.0	(245.6- 760.4)	46.2	(34.7- 57.7)	2.66 (2.47-2.85)	.697 (.669- .725)	5
	Jan. 75	226.4	(106.8- 346.0)	32.6	(21.1- 44.00)	2.56 (2.34-2.77)	.741 (.688- .794)	5
	Apr. 75	305.8	(151.9- 459.7)	36.2	(30.8- 41.6)	2.44 (2.19-2.69)	.683 (.602- .763)	5
	Jun. 75	[481.0	(328.9- 633.1)]	54.5	(41.7- 67.3)	2.79 (2.31-3.28)	.699 (.613- .785)	4
	Sep. 75	302.4	(-41.6- 646.4)	38.8	(26.9- 50.7)	2.74 (2.63-2.85)	.756 (.684- .829)	5
C4	Jun. 74	414.2	(220.5- 607.9)	44.8	(37.5- 52.1)	2.41 (2.07-2.74)	.637 (.529- .744)	5
	Sep. 74	156.8	(83.3- 230.2)	32.8	(26.1- 39.4)	2.51 (1.92-3.11)	.720 (.576- .864)	4
	Jan. 75	222.2	(174.8- 269.6)	31.6	(25.6- 37.7)	1.94 (1.60-2.28)	.557 (.453- .660)	5
	Apr. 75	165.2	(90.9- 239.5)	28.2	(18.8- 37.6)	2.13 (1.86-2.40)	.644 (.614- .674)	5
	Jun. 75	333.4	(149.1- 517.8)	37.8	(21.8- 53.8)	2.06 (1.62-2.50)	.574 (.496- .652)	5
	Sep. 75	174.8	(126.2- 223.4)	34.0	(28.7- 39.3)	2.63 (2.34-2.93)	.747 (.682- .812)	5
C5	Jun. 74	98		31		3.02	.878	1
	Sep. 74	91		30		2.91	.856	1
	Jan. 75	30		11		2.04	.852	1
	Apr. 75	16		7		1.86	.957	1
	Jun. 75	90.0	(19.4- 160.6)	23.8	(13.1- 34.4)	2.50 (1.81-3.19)	.796 (.608- .984)	4

Table 3. (continued)

Station	Date	N	S	H'	J'	n		
C6	Jun. 74	[424.3	(320.6- 528.0)]	[52.0	(41.2-62.8)]	[2.79 (2.03-3.55)]	.706 (.530- .882)	3
	Sep. 74	[1.4 (0.7-	2.1)]	[1.2	(0.6- 1.8)]	[0.14(-0.25-0.52)]	.200(-.355- .755)	5
	Jan. 75	[2.4 (0.5-	4.1)]	[2.2	(0.9- 3.5)]	0.65(-0.09-1.38)	.589(-.121-1.26)	5
	Apr. 75	3.0 (-4.7-	10.7)	0.6	(1.7-(-.51)	0.05(-0.09-0.19)	.074(-.131- .279)	5
	Jun. 75	[15.3 (-25.1-	55.8)]	[5.7	(-3.8-15.1)]	[1.27 (0.63-1.90)]	.838 (.360-1.32)	3
	Sep. 75	[27.0 (15.0-	39.0)]	[9.8	(8.2-11.4)]	[1.92 (1.65-2.19)]	.840 (.773- .907)	5
	Feb. 76	[86.8 (54.6-	118.9)]	[12.8	(7.1-18.4)]	1.28 (0.92-1.65)	.507 (.408- .605)	4
	Sep. 76	[642.8 (147.0-	180.6)]	[35.3	(68.4- 2.2)]	[1.84 (2.46-1.22)]	.527 (.682- .372)]	3
	Jun. 74	660		53		2.58	.649	1
	Jan. 75	111		24		2.35	.740	1
C7	Apr. 75	53.6 (84.1-	23.1)	3.4	(0.3- 6.5)	0.27 (0.03-0.51)	.233 (.405- .062)	5
	Jun. 75	35.0 (-3.1-	73.1)	5.8	(2.4- 9.3)	1.03 (0.18-1.88)	.621 (.226-1.02)	5
	Jun. 74	469.4 (42.6-	896.2)	41.2	(32.9-49.6)	2.41 (1.71-3.12)	.653 (.454- .851)	5
	Jun. 75	152.8 (66.9-	238.7)	27.6	(21.4-33.8)	2.62 (2.19-3.04)	.791 (.680- .902)	5
C8	Jun. 74	439.2 (65.6-	812.9)	41.2	(30.5-51.9)	2.56 (2.19-2.92)	.693 (.578- .807)	5
	Jan. 75	124		25		2.57	.800	1
	Jun. 75	428	(399.0- 457.0)	46.5	(42.3-50.7)	2.79 (2.45-3.13)	.727 (.648- .806)	5
	Feb. 76	367.0		33.0		1.66	.473	2
E1	Jun. 74	[475.7 (332.7-	619.0)]	39.3	(16.8-61.9)	2.24 (0.82-3.65)	.615 (.523- .706)	3
	Sep. 74	504.7 (168.5-	840.9)	26.3	(18.7-34.0)	1.73(-0.17-3.63)	.527 (-.035-1.09)	3
	Apr. 75	118.5		27.5		2.09	.621	2
	Jun. 75	[157.7 (115.3-	200.0)]	28.7	(8.7-48.6)	2.63 (1.79-3.46)	.787 (.703- .870)	3

Table 3. (continued)

Station	Date	N	S	H'	J'	n
E3	Jun. 74	114		2.89	.878	1
	Sep. 74	465.0	(179.4- 750.6)	40.7 (30.3-51.0)	.562 (.245- .880)	3
	Jan. 75	180		1.31	.454	1
	Apr. 75	111		1.62	.560	1
	Jun. 75	188		3.19	.837	1
	Sep. 75	222.3	(50.9- 393.8)	29.3 (18.1-40.5)	.551 (.509- .593)	3
E4	Jun. 74	536.4	(354.7- 718.1)	43.8 (37.9-49.7)	.545 (.459- .630)	5
	Jun. 75	184.5	(-13.3- 382.3)	32.8 (18.9-46.6)	.754 (.596- .911)	4
E5	Jun. 74	507.0		41.5	.534	2
	Sep. 74	377.0		39.0	.558	2
	Jan. 75	225		25	.460	1
	Jun. 74	120		3.13	.854	1
E7	Jan. 75	19.3	(-0.8- 39.3)	2.8 (0.8- 4.8)	.261 (-.033- .555)	4
	Apr. 75	132.8	(-14.8- 280.3)	8.0 (2.3-13.7)	.175 (.092- .258)	4
	Jun. 74	[451.0	(351.4- 550.6)]	[53.0 (44.0-62.0)]	.766 (.715- .817)	4
E8	Jun. 75	[244.8	(138.4- 351.1)]	[31.3 (20.4-42.1)]	.729 (.689- .770)	4
	Jun. 74	519.8	(253.5- 786.1)	42.6 (30.3-54.9)	.640 (-.040-1.32)	5
F3	Jun. 75	372.4	(222.7- 522.1)	39.6 (32.8-46.4)	.578 (.473- .683)	5
	Apr. 75	11		2.15	.977	1
F8	Jun. 74	172.6	(-58.0- 403.2)	30.6 (19.4-41.8)	.698 (.446- .950)	3
	Jun. 75	240.6	(143.4- 337.8)	48.3 (40.3-56.3)	.803 (.629- .977)	3
	Feb. 76					

Table 3. (continued)

Station	Date	N	S	H'	J'	n
F9	Jun. 74	824.0		2.78	.677	2
	Sep. 74	330	60.5	1.68	.504	1
	Jan. 75	36	12	2.10	.843	1
	Jun. 75	469	45	2.30	.604	1
R4	Jun. 74	57	6	1.33	.743	1
	Sep. 74	6	5	1.56	.970	1
	Apr. 75	11	4	1.03	.746	1
	Jun. 75	58.2	(10.3- 106.1)	1.41 (1.11-1.70)	.650 (.570- .731)	5
	Sep. 75	57.5	8.5	1.19	.670	2
R5	Jun. 75	254	21	1.35	.444	1
	Sep. 74	183	19	1.88	.640	1
	Apr. 75	179	9	1.27	.578	1
	Jun. 75	125.8	(35.9- 215.7)	1.51 (0.88-2.13)	.610 (.380- .840)	5
	Sep. 75	148	18	1.73	.597	1
R6	Sep. 74	1058	20	0.59	.198	1
	Apr. 75	1113.5	(820.9-1406.2)	0.37 (0.31-0.43)	.132 (.097- .167)	4
	Jun. 75	855.4	(-22.4-1733.2)	0.84 (0.24-1.44)	.281 (.039- .525)	5
R7	Jun. 74	408	57	3.11	.769	1

in Table 3 to indicate those data sets for which the confidence limits between comparable periods of succeeding years do not overlap. It should be noted that the confidence limits are, in general, quite wide relative to the means. This reflects the substantial macrofaunal variability found between grabs. This variability, and underlying sediment patchiness discussed above, are greater than those we have observed in other inshore areas, i.e., central and western Long Island Sound, Raritan Bay, the New Jersey coast (NMFS, unpub. data). This natural variability tends to obscure any spoiling impacts; effects must be quite pronounced before the CLs for comparable periods become non-overlapping. Thus only 18 of the 100 between-year comparisons in Table 3 showed significant differences based on determinations using 95% CLs.

Eight of the significant differences occurred at the disposal buoy station, C6. Changes in N, S and H' at C6 between Junes and between Septembers were all significant; N and S were also significantly different between Januaries. The between-June changes were decreases from "natural" predisposal populations to the sparse assemblages of newly-dumped spoils, while comparisons of the other months show significant increases in the second year as recolonization proceeded. This sequence is discussed further in the section on recolonization. Significant changes found at other stations are as follows:

Control stations, 2 n mi from disposal buoy - increase in N at A1, and decrease at E1, between Junes; increase in H' and J' at C1 between Septembers.

"Criteria" stations, 1 n mi from buoy - increase in H' between Januaries, and J' between Septembers and between Januaries, at A3; increase in N at C3 between Januaries.

Stations within 1 n mi of buoy - decreases in N and S between Junes at E8.

There may have been meaningful faunal changes at other stations (i.e., C7) which are not listed as significant because samples were not sufficiently replicated to permit 95% CL treatment. Our overall conclusion from Table 3, however, is that spoiling appeared to have no major impacts on community parameters which could be separable from apparently natural fluctuations, at distances ≥ 1 n mi from the disposal buoy. The few control station changes which were found to be significant at 95% CLs showed no clear pattern of increase or decrease. Significant changes at the 1 n mi stations were increases in N , H' and J' whereas spoiling impacts should appear as decreases, at least for H' and J' . Decreases are found where large quantities of spoil are present as at C6 - here the fauna has obviously been directly impacted by burial, and perhaps also by contaminants. At stations close to the disposal point, where apparent decrease in faunal parameters have taken place in the absence of detectable buildup of spoils (as at E8), the possibility of contaminant influences cannot be discounted.

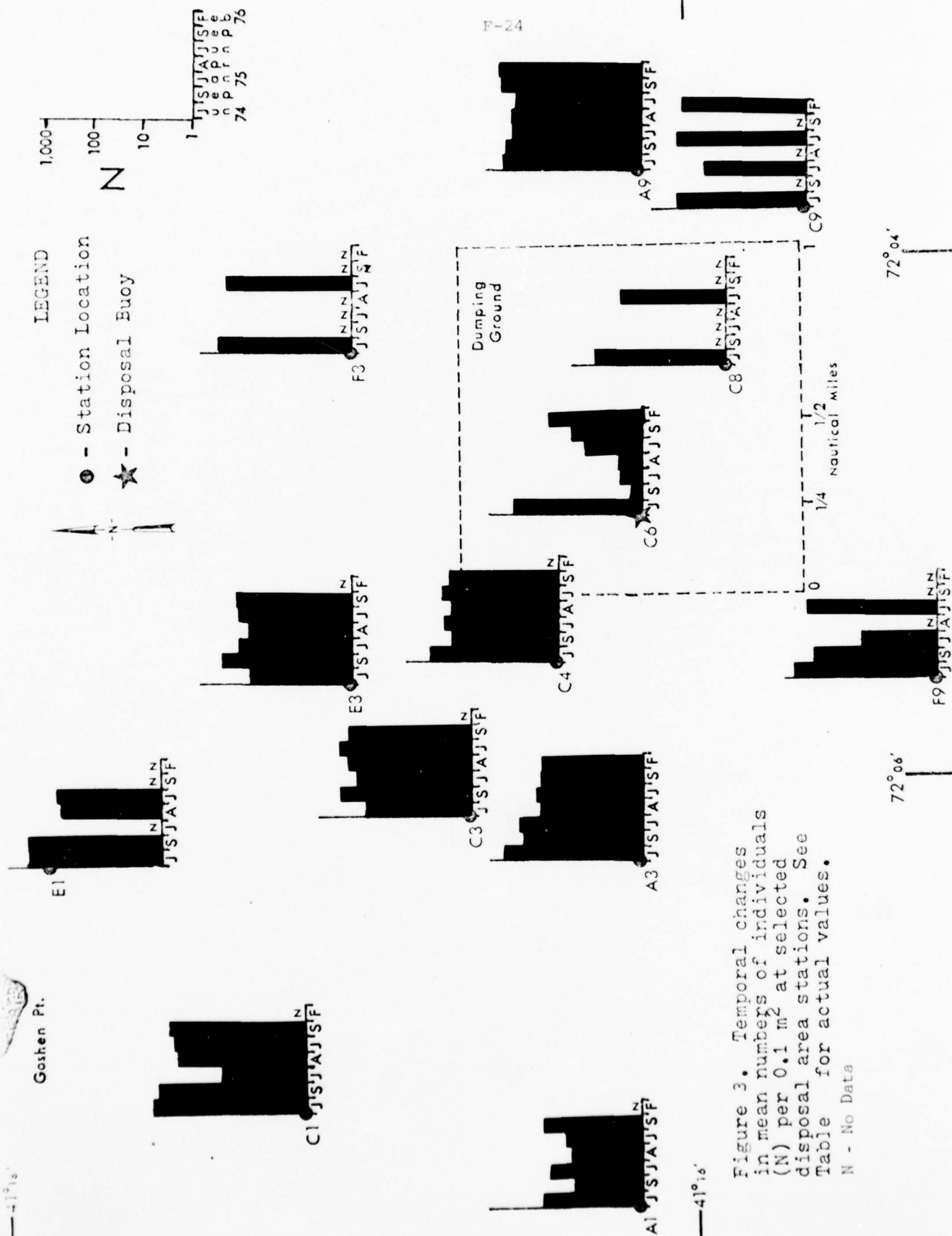
We also examined areal variations in faunal change by dividing the stations listed in Table 3 into four groups, based on distance from the disposal point, and calculating mean N , S , H' and J' \pm 95% CLs for each

group, as well as overall means for all stations, for June 1974 vs. June 1975 (Table 4). Groups are: control (2 n mi from disposal buoy); 1 n mi; 1/2 n mi; and inner ($\leq 1/4$ n mi) stations. Substantial changes between Junes are seen for all groups. As a rule, magnitude of change was in the order: inner $>$ control $>$ 1/2 mi $>$ 1 mi. CLs within any group of stations are wide, again, partly due to the area's faunal patchiness. In no case do within-group CLs fail to overlap between Junes. This is true even for the inner group, where some spoiling impacts are definitely present; in this case impacts are masked by inclusion in the data of station C5, which underwent little change between years. Overall mean N did show significant change between June 1974 and June 1975. Such fluctuations in N are not unexpected; Buchanan, Kingston and Sheader (1974) report natural changes of 100% in faunal density in a long-term monitoring study off Northumberland, England. That the changes at New London are apparently random over space, rather than displaying a gradient of intensity relative to the disposal point, again indicates that spoiling impacts on N, S, H' and J' are small in comparison to natural fluctuations.

Since subtle changes in these parameters may not be uncovered by the 95% CL treatment, they are also presented graphically (Figures 3-5), for 13 representative stations, for possible detection of spoil-related trends. Decreases in faunal density after the onset of spoiling (Figure 3) are clearly evident at C6, and may also be present at C8, A3 and E1. Decreases in species richness (Figure 4) may have occurred

Table 4. Mean values for number of individuals (N), number of species (S), species diversity (H'), and equitability (J') for four groups of stations, as well as overall means (19 stations), from 0.1 m² Smith-McIntyre bottom grab samples. 95% confidence limits given in parentheses; bracketed values denote areas in which confidence limits for succeeding Junes do not overlap. n = number of stations used in calculations.

Station Group	Date	N	S	H'	J'	n				
Control	June 74	688.0	(-1056.0-2432.0)	34.2	(11.0-57.4)	1.80	(0 -3.65)	.53	(-.07-1.13)	3
	June 75	192.7	(- 241.5- 626.9)	25.4	(6.9-43.9)	2.29	(1.26-3.32)	.74	(.27-1.21)	
1 N. Mi.	June 74	448.4	(220.8- 676.0)	47.8	(38.0-57.6)	2.62	(2.30-2.94)	.71	(.61- .81)	7
	June 75	347.9	(272.3- 423.5)	43.6	(37.7-49.5)	2.64	(2.22-3.06)	.70	(.60- .81)	
1/2 N. Mi.	June 74	408.7	(235.8- 581.6)	42.7	(32.7-52.7)	2.46	(2.02-2.90)	.66	(.55- .77)	5
	June 75	231.2	(145.4- 317.0)	36.0	(25.7-46.3)	2.58	(2.11-3.05)	.73	(.62- .84)	
Inner ($\leq 1/4$ N.Mi.)	June 74	422.3	(- 88.0- 932.6)	44.4	(22.0-66.8)	2.59	(1.60-3.58)	.69	(.39- .99)	4
	June 75	46.8	(- 50.0- 143.2)	11.8	(-14.0-37.6)	1.60	(-0.38-3.58)	.75	(.47-1.04)	
Overall	June 74	[470.3	(319.2- 621.4)]	41.7	(37.2-46.3)	2.44	(2.21-2.67)	.66	(.60- .72)	19
	June 75	[239.4	(164.3- 314.5)]	33.1	(26.3-39.9)	2.39	(2.09-2.69)	.73	(.67- .78)	



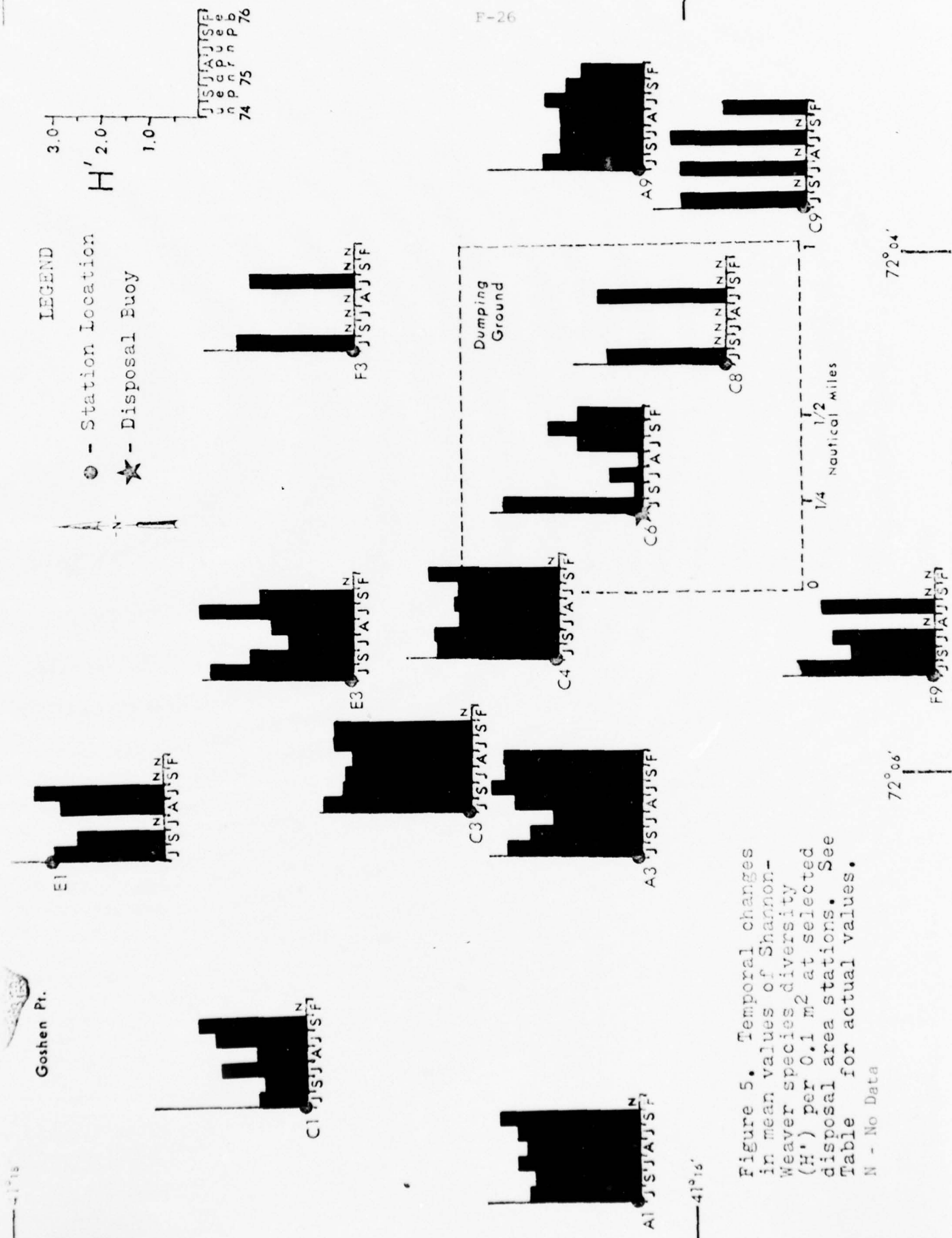


Figure 5. Temporal changes in mean values of Shannon-Weaver species diversity (H') per 0.1 m² at selected disposal area stations. See Table for actual values.

N - No Data

at C8, E1 and F9 in addition to the large changes at C6. Notable declines in diversity are apparent only at C6. The overall pattern, again, is one of random changes, rather than obvious spoil-related effects outside of the immediate disposal area. No striking changes in species diversity (Figure 5) are evident except at the disposal point itself.

Macrofauna Species Composition

Spoiling impacts could also be manifested through changes in the species present or shifts in their relative abundances at affected stations. We examined this possibility in two ways: 1) by comparison of abundances of "common" species (those with mean $N \geq 10/0.1 \text{ m}^2$ at any station for either year) between June 1974 and June 1975; and 2) by comparing cluster analyses (which include all species with mean abundances of approximately 1 or greater per 0.1 m^2 ; see Methods) for the same dates. Densities of the common species, at stations grouped according to distance from the disposal point, are given in Table 5. It can be seen that predisposal assemblages at most stations were dominated numerically by tube-dwelling amphipod crustaceans, including Ampelisca vadorum (the overall dominant), Leptocheirus pinguis, Unciola irrorata and Photis dentata. Most abundant among other taxa were the polychaetes, Clymenella torquata, C. zonalis, Tharyx annulosus and Lumbrineris tenuis, and bivalves, Mytilus edulis and Musculus corrugatus. Earlier surveys in the disposal area had also found assemblages to be dominated by Ampelisca in 1972 and 1973 (Naval Oceano-

graphic Office, 1973; Reid, et al., subm. for publ.)

By June 1975, densities of Ampelisca vadorum were reduced at several stations near the dump site (C8, E4 and A3, in addition to C6 and C7, where nearly all species showed declines). Populations of A. vadorum increased at other stations near the spoil pile, however (C5, F8). It thus appears that overall densities of this dominant amphipod were little affected by spoiling. Conversely, numbers of Leptocheirus pin-
guis, Photis dentata, Mytilus edulis, Musculus corrugatus and Unciola
irrorata decreased throughout the study area. Of the predisposal dominant polychaetes, Lumbrineris tenuis and Tharyx acutus showed no major differences between Junes, while Clymenella torquata and C. zonalis decreased at most stations within 1 n mi of spoiling but not at or beyond 1 n mi.

Other noteworthy decreases between years were for the polychaete, Autolytus alexandri, and caprellid amphipod, Aeginina longicornis (all areas), and polychaete, Harmothoe extenuata, and amphipod Phoxocephalus holbolli (within 1 n mi of spoiling). Prionospio steenstrupi, a polychaete, was the sole species with substantial increases at most stations in June 1975. Although this increased abundance is perhaps a possible indicator of spoiling impacts, an extensive larval set of this species may be another explanation. The same may also apply to Nucula proxima and N. delphinodonta, two bivalves found in the Thames River but not common in the disposal area in June 1974. There may have been an increase in these species, indicative of presence of spoils, at sev-

eral stations by June 1975; the data, however, are not conclusive.

These changes in species composition may imply greater spoiling effects than the analysis of N, S, H' and J' had disclosed. The trend was for taxa which are, on the whole, considered sensitive to stress, such as amphipods, to decrease throughout the 2 mi radius study area, while the generally stress-tolerant polychaetes usually had decreases confined to within one mile of spoiling, showed no changes, or increases. These patterns could be explained by a gradient of spoiling impacts spreading outward from the disposal point. A number of caveats to this interpretation must be mentioned, however. The problems of faunal patchiness and natural yearly fluctuations apply here as they did in the discussion of community parameters. The June 1975 samples were taken only 10 months after spoiling began; some spoiling impacts might only be manifested over a much longer period of time. That Ampelisca, known to be sensitive to some contaminants (Sanders, Grassle and Hampson, 1972), was unaffected, and that both Ampelisca and Leptochaeirus readily recolonized the spoils (see below) is evidence against any gross toxicity of spoil constituents. The strongest statement warranted by the available data is that some changes in species composition, which may be disposal-related, have occurred outside of the immediate spoil pile.

A station-by-station examination of the species changes presented in Table 5 shows the following approximate order of decreases in common species: C6 > C7 > C5, C8 > C4, E4, E8, with no consistent pattern of change at stations further removed from spoiling. The order (greatest

Table 5. Changes in densities of commonly-occurring species at selected stations between June 1974 (values above the line) and June 1975 (values below the line). Stations are listed by distance from disposal point (C6). Values are omitted where densities were below cutoff levels used for cluster analysis; see Methods for further explanation.

SPECIES	1/4 n.mi. from disposal			1/2 n.mi.				1 n.mi.								2 n.mi.							
	C6	C5	C7	C4	C8	E4	E8	F8	A3	A9	C3	C9	E3	F3	F9	A1	C1	E1	R4	R5			
Nephtys incisa	3.0 1.3	3.0 0.8	5.0 2.8	2.0 1.4	4.0	2.7 0.8	0.7 2.0		2.3	6.3 2.0		0.7 1.3		5.0			1.0		4.0 0.6	20.0 12.8			
Autolytus verrilli	1.7			0.8		10.3			4.3	1.0	1.8					1.0							
Scalibregma inflatum	5.3	1.0	2.0	8.0 4.0	3.0	15.3 1.8	2.3 8.3		4.8 1.0	5.0 21.0	9.8	1.7 12.5		2.3 5.0	4.0								
Clymenella torquata	23.7	2.0	17.0	15.0 8.0	7.3 0.8	11.7 5.5	0.7 8.0		2.5	17.7 36.6	3.4 0.8	6.5		13.7 13.4						1.0			
Clymenella zonalis	11.0	10.0	20.0	9.6	11.0	4.0	10.3 2.5	10.7 8.7	15.8 5.5	2.7	8.2 3.3	4.0 4.8	6.0	9.3 4.0	8.0 9.0		0.8 5.0			1.0			
Harmothoe extenuata	8.0	5.0 1.0	15.0	3.8 1.8	9.0	3.0 2.5	9.7 7.3	0.7 7.3	7.3 5.5	3.3 1.8	7.0 11.0	2.0 1.8	6.0 16.0	5.3 3.8	47.0 6.0		3.8 23.0	1.7					
Polydora caulleryi	0.7				0.8		0.7 1.7				1.4 3.5			15.0 5.0			4.0	1.3					
Polydora ligni					0.8					1.3									6.0 27.8	21.0 12.6			
Capitella capitata			0.4														18.3		7.0				
Potamilla reniformis	5.7	1.0	7.0	2.0 0.8	1.3	0.8	2.0 2.0	0.7 2.0	2.5 3.8	1.7	3.0 3.8	1.0 5.3	8.0 19.0	0.7 1.6	1.0 1.0								
Spiophanes bombyx		1.0		7.2 0.8	1.5		9.0 8.7		126.5	8.4	6.6 2.0	1.3	1.0 3.0	3.8	6.0 6.0	21.4	49.0	7.3					
Tharyx acutus						0.7	8.0 12.0		3.0 3.0	1.0 4.2	4.4 12.8		1.0 3.0	1.3 6.2	53.0 42.0	1.2	5.0		0.4	1.0 1.2			
Tharyx annulosus	17.7	2.0 3.8	40.0 0.2	9.8 5.0	5.5 9.8	6.0 14.3	54.3 22.5	78.0 13.3	37.5 9.0	6.3	20.4 52.5	17.0 68.5	36.0	15.7 28.0	275.0 185.0	0.6	1.0 2.0	8.0 4.0					
Prionospio steenstrupi		1.0	7.0	0.8 40.2	1.3	0.7 35.3	19.0		3.3 1.5	22.0	1.0 10.8	39.3	3.0	5.4	5.0 102.0		2.3						

Table 5. (continued)

SPECIES	$\frac{1}{4}$ n.mi. from disposal			$\frac{1}{2}$ n.mi.			1 n.mi.			2 n.mi.			R5						
	C6	C5	C7	C4	C8	E4	E8	F8	A3	A9	C3	C9		E3	F3	F9	A1	C1	E1
<i>Lumbrineris tenuis</i>	11.7	2.0 6.3	19.0 0.4	0.6 7.6	13.8 7.4	4.0 10.0	25.3 11.3	5.3 4.0	9.8 4.8	7.3 1.8	1.2 54.3	13.3 18.3	5.0 13.0	29.3 22.8	15.0 9.0			12.0 5.0	0.4 0.8
<i>Polycirrus eximius</i>	2.3				0.8	0.7 1.0	1.0 1.3	1.0	1.0	1.0	1.6 23.5	26.5	9.0 7.0	1.3 1.2	13.0 11.0	1.0	1.4 3.0	32.3 41.0	1.0
<i>Ninoe nigripes</i>	7.0	2.0 6.8	6.0 0.4	8.2 15.6	6.3 8.6	8.3 6.0	3.7 13.5	0.7 0.7	1.5 2.8	8.0 8.4	2.0 2.8	6.0 9.5	4.0	5.3 5.2					0.8
<i>Aricidea jeffreysii</i>		1.0		0.8				1.3			0.8		2.0		101.0 13.0	1.6	2.8 1.0	3.0 12.3	2.0 2.6
<i>Ampharete arctica</i>	3.3	2.0 0.8	4.0	4.4 2.0	4.0	3.0 4.3	6.0	3.7 9.0	5.5 4.3	2.0 3.0	3.8 7.8	2.3 3.5	5.0	2.3 3.4	17.0 4.0		1.2	2.0 3.0	
<i>Mitrella lunata</i>	4.7					1.0	0.7	1.7	1.5 3.0	3.0	1.6 3.8		11.0		1.0		1.4 3.7	2.7	
<i>Pandora gouldiana</i>	1.7		2.0				1.0	0.7		1.0		0.7 1.0			23.0			1.0	
<i>Lyonsia hyalina</i>	3.0		8.0	5.4 1.2	2.8	3.3 0.8	3.0	1.3 3.0	4.0 1.3	2.7		0.8	1.0 1.0	3.0	22.0 3.0	1.0	1.6	3.0	
<i>Crenella glandula</i>			1.0		1.0 1.8	0.7 0.8	0.7 0.8			1.7		9.0	8.0	7.0	14.0 4.0				
<i>Nucula proxima</i>	13.3 8.7	8.0 3.8	6.0 28.7	9.0 4.8	5.0 17.2	2.7 3.0	6.7 7.5	1.0 4.7	3.3 24.5	1.7 7.8	2.4 2.8	2.0 1.5	1.0	1.0 1.4	7.0 1.0			32.0 1.4	171.0 76.4
<i>Nucula delphinodonta</i>				11.6	6.6	1.7	4.5			7.7 3.8	26.0	3.3		4.0					
<i>Mytilus edulis</i>	3.3	1.0	4.0	6.0	2.8	3.7	12.0	3.7 2.7	36.5 1.8		4.6 1.3	10.7	5.0		65.0 1.0	1.8	1278.4 248.0	27.3	

Table 5. (continued)

SPECIES	$\frac{1}{4}$ n.mi. from disposal			$\frac{1}{2}$ n.mi.				1 n.mi.				2 n.mi.				R5				
	C6	C5	C7	C4	C8	E4	E8	F8	A3	A9	C3	C9	E3	F3	F9		A1	C1	E1	R4
Musculus corrugatus	10.0	3.0 1.5	46.0	12.2 3.2	6.8	9.0	31.7 2.3	1.0 3.7	19.5 1.0	7.7 2.0	2.2 4.5	10.7 3.5	2.0 1.0	5.0 1.0	12.0 5.0		21.8			
Cerastoderma pinnulatum	3.7	5.0	9.0	4.4	2.3 2.6	3.0 1.3	9.0 5.8	1.3 6.7	10.8 1.8	2.3	1.8 3.3	3.3 11.5	3.0	2.0 1.4	14.0 8.0					
Astarte undata	1.0	2.0	1.0	1.2 5.0	1.0 1.2	1.3 0.8	3.7 3.5	0.7 0.7		5.0 7.6	2.0 9.8	2.0 8.8	5.0	9.0 4.6	4.0 3.0	1.6	1.4 1.0			
Aeginina longicornis	6.0	2.0 0.8	9.0 0.2	3.8 2.6	8.8	5.0	12.0	1.7 11.0	8.5 4.0	4.0 0.8	2.2	0.7	2.0 1.0	0.7 1.0	6.0		12.6			
Photis dentata	2.3	0.8	22.0	7.8 8.4	15.5	35.7 7.3	7.0 1.5		9.3		1.5	19.3 10.5	1.0	33.7 7.0		11.4				
Unciola irrorata	13.3	4.3	13.0	10.2 4.8	7.5	17.7 3.0	13.7 7.8	3.0 3.0	18.3 1.0	11.7 7.6	4.0 1.0	4.0 2.8	5.0	10.7 6.8	4.0 1.0	4.6 0.6	5.4	1.0 4.8		
Leptocheirus pinguis	47.7	2.0 2.0	76.0 0.2	36.6 14.2	61.5 2.4	88.0 7.8	50.7 6.3	5.3 12.7	81.8 3.5	62.3 2.8	4.6 7.5	27.0 9.3	3.0	61.0 17.8	2.0	1.4				1.0
Phoxocephalus holbolli	1.7	4.0	2.0	2.4	4.5	0.8	9.7	1.7 2.0	7.5 3.3		7.0	1.3 1.0	10.0	1.0 1.0	21.0 4.0					
Ampelisca vadorum	141.3	18.0 33.0	257.0 0.4	201.6 180.6	160.5 9.4	278.3	93.7 77.5	10.0 39.3	118.5 5.0	221.7 158.4	25.2 126.0	66.0 99.3	3.0 3.0	171.0 191.2	7.0 3.0	4.2	35.2	1.0		
Amphipholis squamata	0.7	2.0		0.8			0.7	0.7			1.0 1.5			2.7	15.0 2.0		1.3			

effects at 1/4 n mi were at C7, and at 1/2 n mi, C8) suggests that at a given distance from the release point, changes are greatest in a southeast direction. This agrees with the direction of net bottom currents measured for the disposal area (Morton, Cook and Massey, 1975; Section C of this report).

A second means of comparing species composition for June 1974 vs. June 1975 is through cluster analysis. Q-mode dendrograms for these two sampling periods are presented in Figure 6. The June 1974 (pre-dredging) dendrogram shows species composition at Station C6, the future disposal point, to be closely related to that at a number of other stations. C6 clusters with C7, E4, C8, A3 and E8 at a percent similarly (PS) level of $>70\%$, and with C4, F3, C3 and C5 at 60% . By comparing Figure 6 with Table 5, it can be seen that dominance by amphipods was a major contributor to the similarity of all these stations except C3 and C5. Stations with high PS's also tended to be physically closer together; the control stations, at 2 n mi radii from C6, had among the lowest PS's, generally followed by 1 n mi stations.

Comparison with the June 1975 dendrogram shows, as expected, a large change in species composition at C6 relative to that at other disposal area stations. In June 1975, C6 had a PS of $>40\%$ to only one other station, C7, also located in the spoil pile. C6 and C7 were quite dissimilar to the remaining stations joining the larger cluster at the 24% level. Spoiling impacts at other stations might thus appear as a weakened affinity to the central cluster which had been present

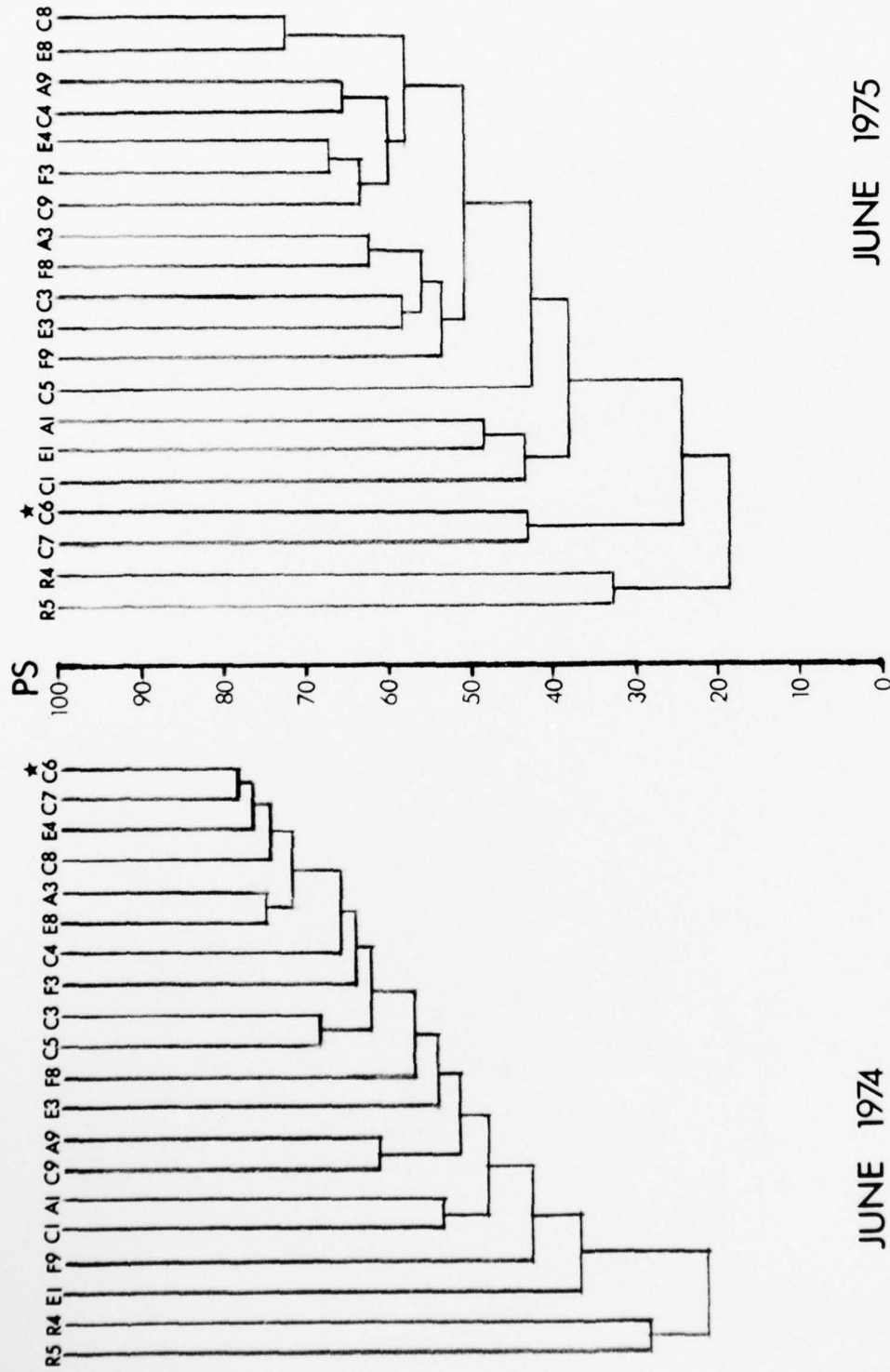


Fig. 6. Q-mode dendrograms showing percent similarity (PS) between stations, based on Czekanowski coefficient and group-average sorting, for June 1974 (predisposal) and June 1975. ★ = disposal point.

in June 1974, and perhaps also as some degree of similarity to the altered assemblages at C6 and C7. Station C5, and possibly A3, do show a decreased similarity to the central cluster; in fact, there is a slightly lower overall similarity among stations within this cluster. No tendency to approach the species composition of C6 and C7 is evident, however. On the whole, species compositions as reflected by order of clustering and degree of similarity do not appear greatly changed between June 1974 and June 1975.

Recolonization

Recolonization of the spoil pile has been followed through September 1976, 25 months after onset and 14 months after termination of spoiling. Station C6, in the approximate center of the spoil pile, has been sampled eight times. Sampling dates, and mean values obtained for N, S, H' and J', are given in Table 3 and Figure 7. As Table 3 indicates, June 1974 (predisposal) samples at C6 were representative of the New London area in terms of N, S, H' and J'. Species composition was also typical of the area, with dominance by tube-dwelling amphipods. Figure 7, tracing the temporal changes in N, S and H' at C6, reveals that all these faunal parameters had dropped precipitously by September 1974. Samples taken at that time consisted of freshly-deposited spoils containing few or no macrofaunal organisms. There was little change in the fauna at C6 through the April 1975 samples, but an upward trend in N, S and H' was apparent by June 1975, and by September 1975 all values were significantly higher (95% CLs) than

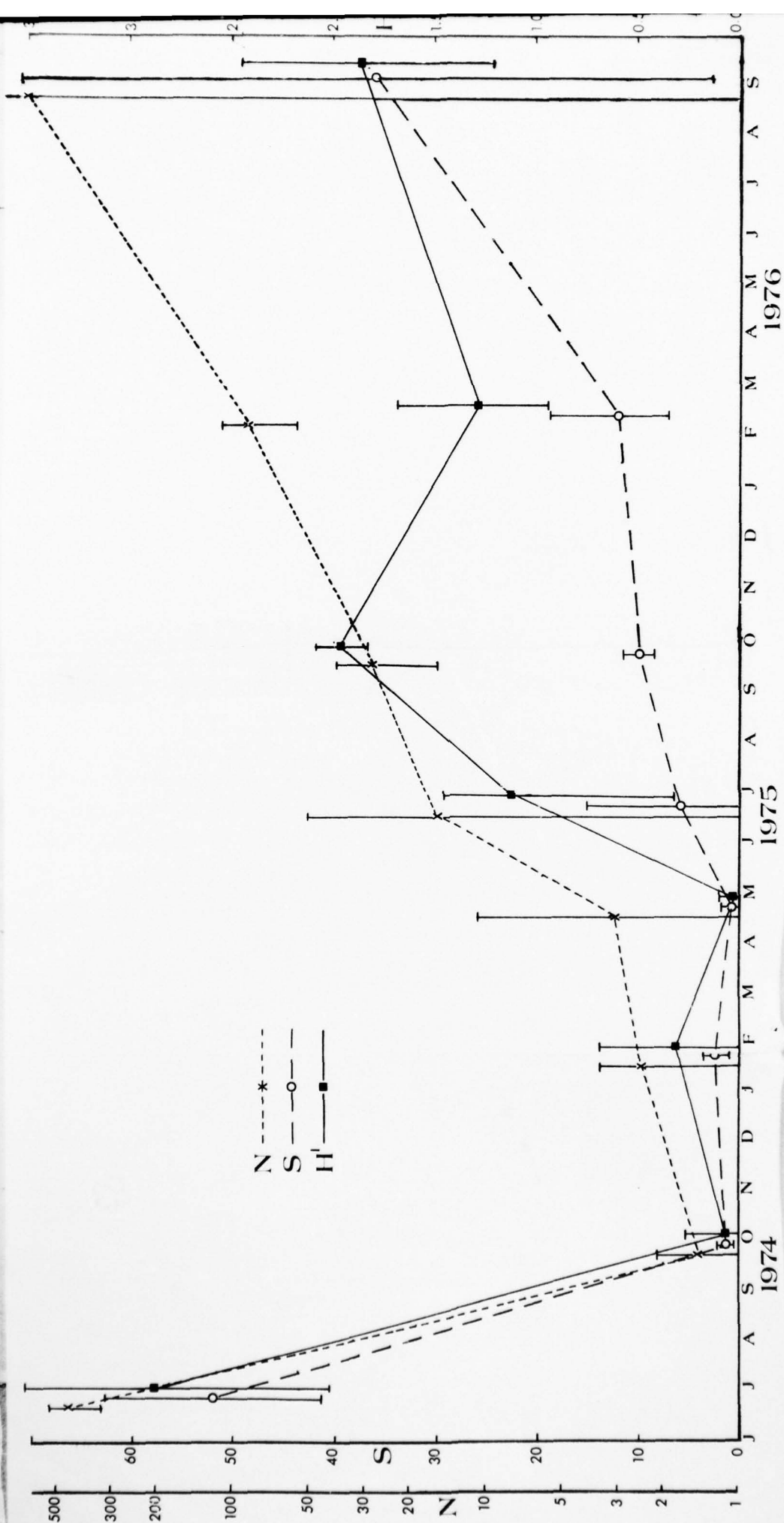


Fig. 7. Means and 95% confidence limits for numbers of individuals (N), numbers of species (S) and species diversity (H') at the original disposal point (C6), July 1974 to September 1976; spoiling began in August 1974.

for the preceding September, January or April. Recolonization proceeded through winter 1975-76. By September 1976, mean N was greater than it had been in the predisposal samples.

The recolonizing organisms were dominated by Ampelisca vadorum and Leptocheirus pinguis, as predisposal assemblages had been. SCUBA surveys made in October 1975 revealed that populations of tube-dwelling amphipods were reestablished over much of the spoil pile, as well as at the disposal buoy). Reappearance of these species is significant for a number of reasons. First, as noted above, they are thought to be sensitive to environmental contamination. Their presence on the spoils may be taken as evidence that contaminants are at tolerable levels. Second, amphipod tubes have been demonstrated to stabilize sediments (Mills, 1967). They may thus inhibit erosion of the spoil pile while modifying the benthic environment to promote further recolonization.

Thirdly, although ampeliscid amphipods have been shown to appear early in a sequence of species colonizing new environments (Fisher, 1973), they are also dominant members of what appear to be "Climax" assemblages in silty-sand sediments of eastern Long Island Sound (Reid, et al., subm. for publ.) as well as in neighboring Block Island Sound (Steimle, et al., 1976), Rhode Island Sound (Saila, Pratt and Polgar, 1972), and the northeastern continental shelf in general (Pratt, 1973). That these species are early colonizers at New London may indicate that recovery toward predisposal conditions will be relatively rapid. In

Rhode Island Sound, ampeliscids had recolonized an edge of the spoil pile (which may have been relatively free from active spoiling for as much as three years) while spoiling was still ongoing elsewhere in the disposal area (Saila, Pratt and Polgar, 1972). A year later, on the amphipod-dominated fringe of the spoils, "many spoil samples contained 30-40 species and are not separable from natural bottoms on the basis of species numbers alone" (Pratt, Saila and Sissenwine, 1973). We expect a similar pattern of recolonization for the New London spoils. The Naval Oceanographic Office (1973) study revealed a marked recovery of benthic fauna within six months after termination of a smaller disposal project at New London. Ampeliscid amphipods especially had re-established themselves, often in greater numbers than were found prior to disposal.

Finally, both Ampelisca and Leptocheirus are important in the diets of several of the area's more valuable finfish species, such as winter flounder and scup (Section G of this report, and Steimle, et al., 1976). The early reappearance of these forage species implies that overall productivity of the New London area would not show lasting effects of the disposal of the first increment of Thames River spoils. An interesting finding from Section G of this report was the heavy predation by demersal fish on a species of Gammarus (Amphipoda) which was not commonly collected in benthic sampling. This species may have been one which is more pelagic than most of the genus (Bousfield, 1973), or perhaps the fish sampled had also been feeding elsewhere.

SUMMARY

1. Sediments and benthic macrofauna of the Thames River and New London disposal area were sampled on an approximately quarterly basis between June 1974 and September 1976 to monitor spread of spoils and impacts on benthic communities.
2. Four stations, all $\leq 1/2$ n. mi of the designated disposal point, consistently had increased silt-clay percentage, indicative of spoil buildup, after disposal began. Use of sediment data to delimit the spoil pile was made difficult by sampling variability and spatial heterogeneity of the sediments.
3. Macrofauna community structure analysis showed large changes in faunal densities over space and time, with slightly smaller fluctuations in numbers of species, species diversity and equitability. Outside of the immediate spoil pile, changes appeared random relative to distance from the disposal buoy. It was thus concluded that spoiling impacts were small in comparison to possible natural fluctuations and spatial heterogeneity for the study area as a whole.
4. Observed changes in species composition may indicate slightly greater spoiling impacts than were revealed by analysis of N, S, H' and J'. Changes in species composition could thus be a more sensitive measure of spoiling effects.

5. Significant recolonization of the spoil pile was seen within 13 months of the onset of disposal. Numbers of species and individuals continued to increase through the September 1976 sampling. Dominant recolonizing forms were those which had characterized predisposal communities in the area. Fairly complete and rapid recovery from the first increment of spoiling was predicted.

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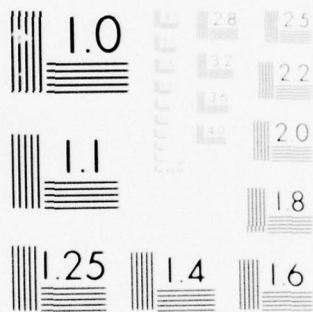
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G. DEMERSAL FINFISH OF DISPOSAL AREA
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INTRODUCTION

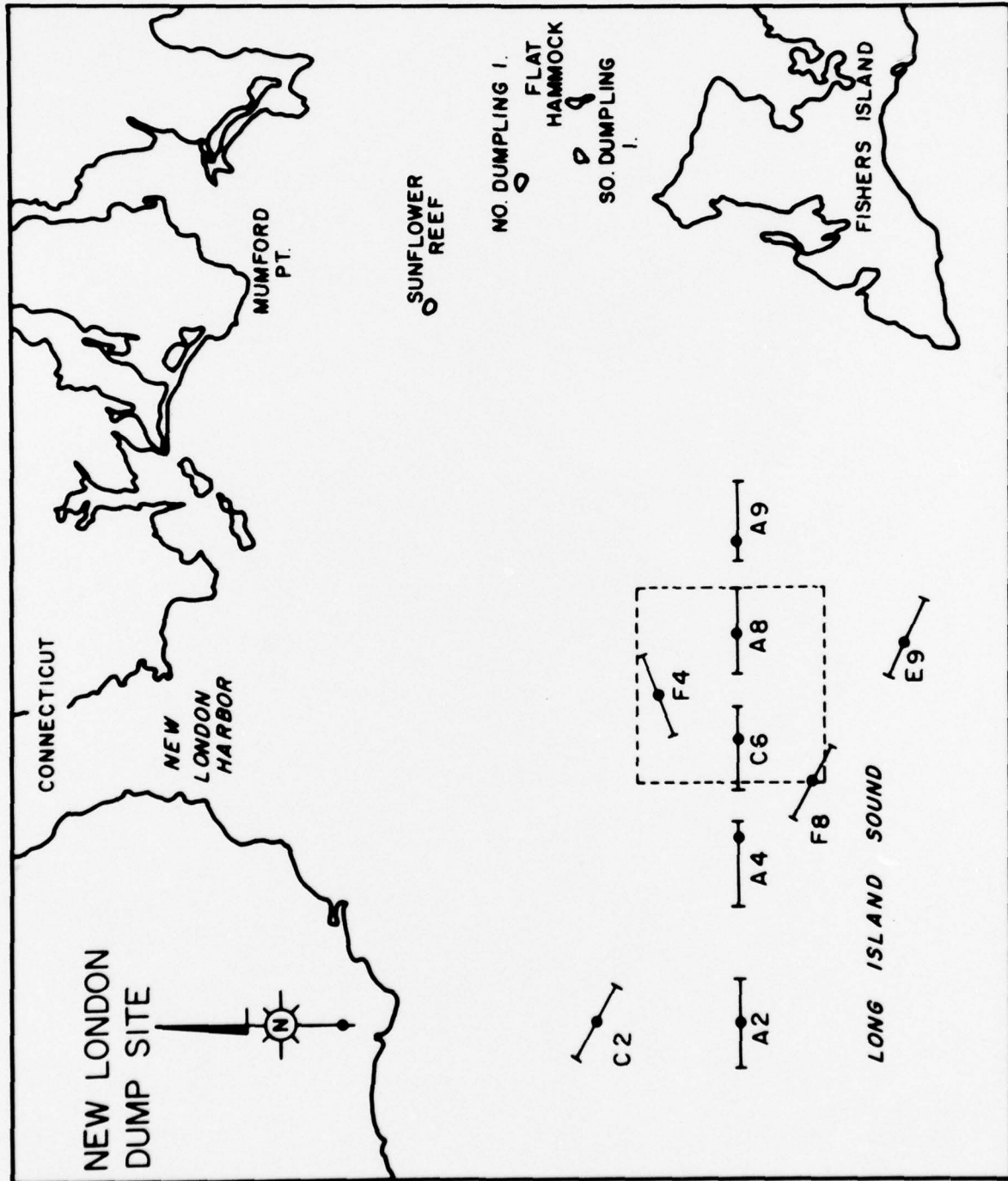
The Fisheries Laboratory of the New York Ocean Science Laboratory, under contract from (NOAA) National Oceanic and Atmospheric Administration, sampled the demersal fish population in the area of the New London Disposal Site during four cruises: before disposal in July, 1974, during disposal in October, 1974, and again in February, 1975, and after completion of the disposal in August, 1975. Sampling was conducted at nine stations chosen from those previously used for sampling of benthic invertebrates (Figure 1). The selection of these stations was based upon their proximity to the site, bottom type, water depth and direction of prevalent tidal currents.

METHODOLOGY

Sampling of demersal fish populations at the New London Dump Site was conducted on four cruises: 18-19 July, 1974; 9-11 October, 1974; 18-19 February, 1975; and 4-6 August, 1975.

Trawls were made using a 35' lead rope otter trawl (2" s.m. body, 1" s.m. cod end with 1/4" s.m. liner) at nine stations (Figure 2). The duration of tows was for a period of fifteen minutes when conditions permitted. Triplicate tows were made in July, 1974 at Stations A4, C2, C6; in October, 1974 at A2, A9 and C6; and in February, 1975 at Stations A2 and C2. In August triplicate tows were made only at Station A9. Duplicate trawls were

Figure 1



Station Locations

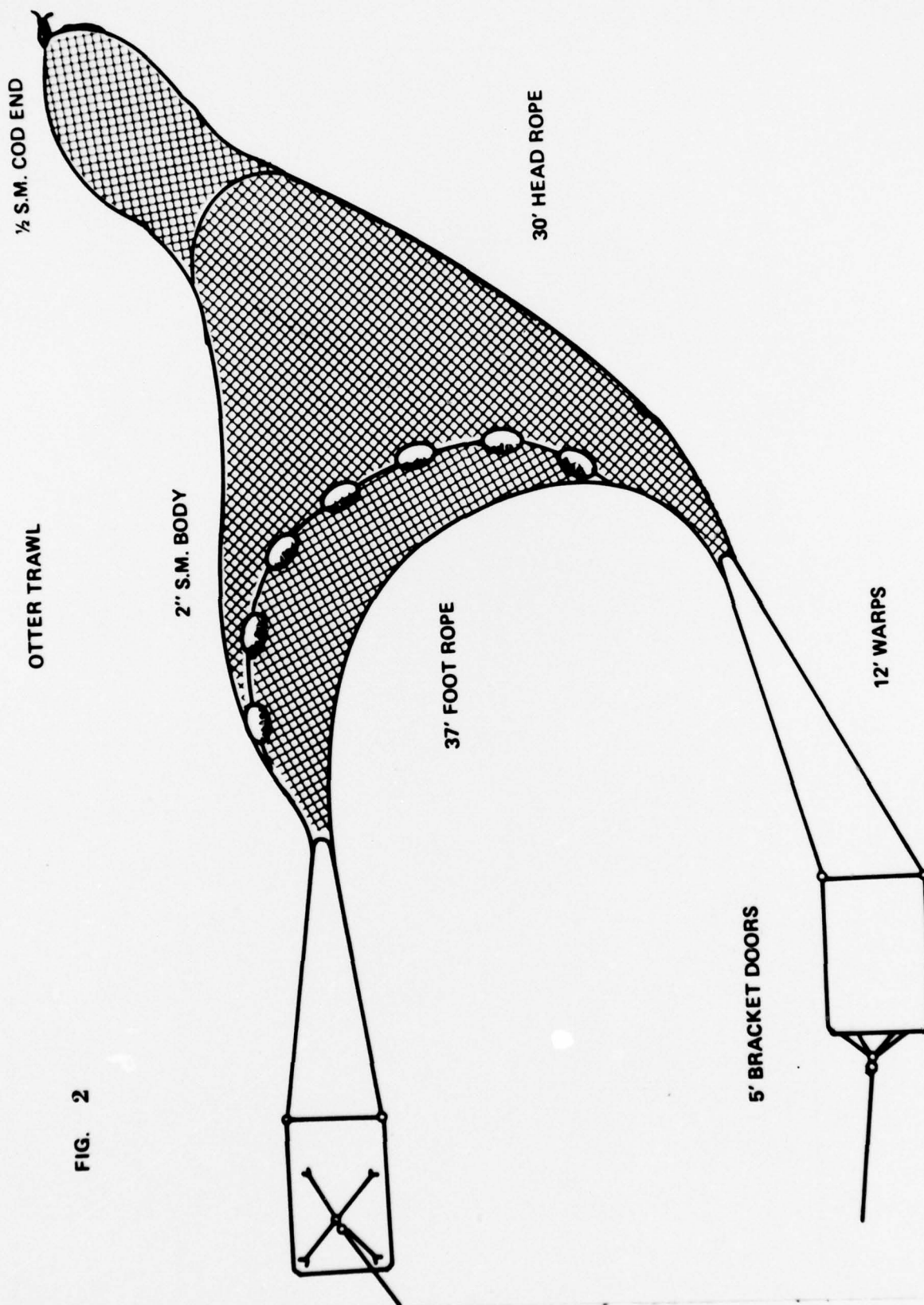


FIG. 2

made in July, 1974 at Station A4; in October, 1974 at C2; and in February, 1975 at C6.

Physical parameters recorded before each tow included water column depth, bottom temperature, and bottom salinity (Table 1). Depth and temperature were obtained via bathythermograms.* Water samples were taken using a 5 liter Niskin bottle and salinities measured with an induction salinometer.**

All fish captured were identified and enumerated. A representative sample of each species was kept for laboratory analysis, which included measurements of standard length (mm) and total weight (g) for each fish. Gonads were excised, weighed and the gametes were examined under a microscope to determine sex. Stomachs were removed and contents were identified to the lowest possible taxa and each taxonomical group was weighed (Figures 3-16). The stomach contents were removed from those fish caught at the following stations: A9 which is east of the Disposal Site, C6 which is the Disposal Site and C2 which is west of the disposal area (Figure 1). Ages were calculated by counting annual rings on scales and/or otoliths depending upon species (Table 10). Male to female ratios were also recorded (Table 2).

RESULTS AND DISCUSSION

The mean total catch was 1,577 fish which were collected over the four

* Bathythermograph manufactured by Belfort Instrument Co.

** Induction salinometer manufactured by Beckman Instruments Co.

G-5

Table 1

Mean Bottom Depth, Temperatures and Salinity

N.S. = Not sampled

July 1974

Station	Depth(m)	Bottom Temperatures (°C)	Bottom Salinity (°/∞)
A2	24	16.8	30.554
A4	20	17.1	30.374
A8	22	17.5	30.274
A9	18	17.7	30.104
C2	18	17.6	30.050
C6	22	17.0	30.386
E9	23	16.3	30.898
F4	16	17.9	30.212
F8	20	17.2	30.193

October 1974

Station	Depth(m)	Bottom Temperatures (°C)	Bottom Salinity (°/∞)
A2	18	16.3	31.634
A4	18	16.6	31.252
A8	17	16.8	31.308
A9	16	16.6	31.440
C2	11	16.2	31.298
C6	19	16.6	31.326
E9	N.S.	N.S.	N.S.
F4	14	16.4	31.448
F8	21	16.6	31.182

Table 1 (cont.)

February 1975			
Station	Depth(m)	Bottom Temperatures (°C)	Bottom Salinity (°/‰)
A2	17	3.5	31.034
A4	19	3.4	30.967
A8	18	4.0	30.969
C2	N.S.	N.S.	30.920
C6	18	3.5	30.903
E8-E9	20	3.4	30.859
F4	10	3.9	30.965
F8	19	3.0	30.743

August 1975			
Station	Depth(m)	Bottom Temperatures (°C)	Bottom Salinity (°/‰)
A2	20	18.8	30.484
A4	22	17.0	29.758
A8	20	18.5	30.722
A9	20	19.0	30.619
C2	15	20.0	30.733
C6	18	20.0	30.618
E9	25	17.5	31.055
F4	18	18.9	30.769
F8	22	18.1	29.770

Table 2

Male:Female Ratios for 13 Species
collected at 3 stations

Species	Totals per station			All Stations Σ
	A9	C2	C6	
<i>Tautoga onitis</i>	0:2	1:0		1:2
<i>Stenotomus chrysops</i>	10:6	8:2	3:1	21:9
<i>Raja erinacea</i>	1:8	1:4	3:8	5:20
<i>Tautogolabrus adspersus</i>	2:1	3:5	3:1	8:7
<i>Paralichthys dentatus</i>	1:3			1:3
<i>Scophthalmus aquosus</i>	12:11	1:2	4:2	17:15
<i>Pseudopleuronectes americanus</i>	22:20	14:13	19:31	55:64
<i>Prionotus carolinus</i>	1:1	4:1	0:1	5:3
<i>Monacanthus hispidus</i>		1:0		1:0
<i>Menidia menidia</i>		2:2		2:2
<i>Hemitripterus americanus</i>		0:1		0:1
<i>Trinectes maculatus</i>			1:0	1:0
<i>Myoxocephalus octodecemspinosus</i>	0:2			0:2
Σ	49:52	35:30	33:44	117:128

samplings (Tables 3-7). This value included duplicate and triplicate tows. The amount of fish caught in July before disposal activities began, comprised 59.4% of the total catch. The catch in October, 1974 and February, 1975, during the disposal, represented 16.6% and 6.6%, respectively. The August, 1975 sampling, which took place after completion of disposal, accounted for 17.6% of the catch, a slight recovery.

The seasonal and spatial distribution that is discussed represents a mean value in order to standardize the results obtained from single and replicate trawls.

The mean numbers of fish caught in July, 1974 and August, 1975, were 835 and 393, respectively. Therefore, a substantial portion (78%) of the fish caught were taken either before or after completion of disposal activities. For all fish captured on the New London Disposal Site Project (Table 7, Figure 17), the largest value of fish (N=257) was collected at Station C6 in July, 1974. This station was later the center of disposal activities. Sampling at this station during disposal resulted in no yield. Overall comparisons of catches at all stations from July, 1974 and August, 1975 resulted in the August catch being less with the exception of A8. The station yielding the least amount of fish over the entire sampling was E9.

The mean catch per hour for nine species of fish can be seen in Table 8.

Table 3

Mean Number of Fish Captured Trawling
July 1974

	A2	A4	A8	A9	C2	C6	E9	F4	F8	Total
<i>Pseudopleuronectes americanus</i>	4	0	69	38	34	249	0	77	112	583
<i>Tautoglabrus adspersus</i>	1	13	0	2	3	2	0	41	15	77
<i>Stenotomus chrysops</i>	16	6	3	12	15	0	1	1	4	58
<i>Scophthalmus aquosus</i>	-	2	13	27	1	1	1	5	9	59
<i>Tautoga onitis</i>	9	4	0	1	1	1	0	3	0	19
<i>Raja erinacea</i>	7	1	0	1	2	2	0	3	0	16
<i>Prionotus carolinus</i>	0	0	0	0	0	1	0	3	4	8
<i>Paralichthys dentatus</i>	0	1	0	1	0	1	0	3	4	10
<i>Urophycis chuss</i>	0	1	0	0	0	0	0	1	0	2
<i>Mustelus canis</i>	1	0	0	0	0	0	0	0	0	1
<i>Paralichthys oblongus</i>	0	0	0	0	0	0	1	0	0	1
Mean Total Fish Captured per Station	38	28	85	82	56	257	3	137	148	834

Table 4
Mean Number of Fish Captured
October 1974

	A2	A4	A8	A9	C2	C6	F4	F8	Total
<i>Stenotomus chrysops</i>	25	0	5	25	0	1	10	0	66
<i>Pseudopleuronectes americanus</i>	1	28	2	2	5	1	16	3	58
<i>Scophthalmus aquosus</i>	6	0	10	2	0	1	5	20	44
<i>Tautoglabrus adspersus</i>	0	1	9	1	15	0	41	0	67
<i>Peprilus triacanthus</i>	6	1	0	0	0	0	0	0	7
<i>Prionotus carolinus</i>	0	1	0	0	0	0	1	2	4
<i>Tautoga onitis</i>	1	0	0	0	0	0	0	1	2
<i>Monacanthus hispidus</i>	0	0	1	0	5	0	1	0	7
Mean Total Fish Captured per Station	39	31	27	30	25	3	74	26	255

Table 5

Mean Number of Fish Captured
February 1975

	A2	A4	A8	A9	C2	C6	E8-E9	F4	F8	Total
<i>Pseudopleuronectes americanus</i>	0	0	25	12	0	0	1	16	1	55
<i>Urophycis chuss</i>	0	0	2	0	0	1	0	0	0	3
<i>Myoxocephalus octodecemspinosus</i>	0	0	4	2	0	0	1	3	8	18
<i>Raja erinacea</i>	0	0	1	2	0	3	0	1	0	7
<i>Macrozoanceus americanus</i>	0	0	1	0	0	0	0	0	0	1
<i>Alosa pseudoharengus</i>	0	0	1	0	0	0	0	0	0	1
<i>Hemitripterus americanus</i>	0	0	1	0	0	0	0	0	0	1
<i>Myoxocephalus aeneus</i>	0	0	0	0	0	0	0	1	0	1
<i>Photis gunnellus</i>	0	0	0	0	0	0	0	0	1	1
<i>Scophthalmus aquosus</i>	0	0	0	2	1	0	0	0	0	3
<i>Menidia menidia</i>					3					3
Mean Total Fish Captured per Station	0	0	35	18	4	4	2	21	10	94

Table 6
Mean Number of Fish Captured
August 1975

	A2	A4	A8	A9	C2	C6	E9	F4	F8	Total
<i>Pseudopleuronectes americanus</i>	0	2	149	23	1	63	0	7	37	282
<i>Scophthalmus aquosus</i>	0	2	11	5	0	4	0	2	4	28
<i>Raja erinacea</i>	0	1	13	6	1	3	2	0	5	31
<i>Paralichthys dentatus</i>	0	0	0	1	0	0	0	0	0	1
<i>Stenotomus chrysops</i>	0	0	9	4	0	2	0	3	2	20
<i>Prionotus carolinus</i>	0	0	8	1	0	1	0	2	0	12
<i>Tautoglabrus adspersus</i>	0	0	2	0	6	3	0	0	1	12
<i>Peprius triacanthus</i>	0	0	0	0	0	1	0	1	0	2
<i>Tautoga onitis</i>	0	0	1	0	0	0	0	1	0	2
<i>Paralichthys oblongus</i>	0	0	1	0	0	0	0	0	0	1
<i>Anchoa mitchilli</i>	0	0	0	0	0	0	0	1	0	1
<i>Alosa aestivalis</i>	0	0	0	0	0	0	0	1	0	1
Mean Total Fish Captured per Station	0	5	194	40	8	77	2	18	49	393

Table 7

Mean Total Catch Per Station Per Sampling
N.S. = Not sampled

Station	July 1974	October 1974	February 1975	August 1975	Total
A2	39	39	0	0	78
A4	28	31	0	5	64
A8	85	27	35	194	341
A9	82	30	18	40	170
C2	56	25	4	8	93
C6	257	3	4	77	341
E9	3	N.S.	2	2	7
F4	137	74	21	18	250
F8	148	26	10	49	233
Total	835	255	94	393	1577

The species included are winter flounder, *Pseudopleuronectes americanus*; scup, *Stenotomus chrysops*; cunner, *Tautoglabrus adspersus*; windowpane flounder, *Scophthalmus aquosus*; tautog, *Tautoga onitis*; little skate, *Raja erinacea*; Northern searobin, *Prionotus carolinus*; summer flounder, *Paralichthys dentatus* and longhorn sculpin, *Myoxocephalus octodecemspinosus*. Fish species caught less than ten times during the entire sampling are not included in the catch per hour values but are reported in Table 9.

1. Abundance

Winter flounder

The winter flounder was the most abundant species of fish captured and it does comprise a commercial fishery in the area as well as a sport fishery in the inshore areas of Block Island Sound. It comprised as much as 71.8% of the mean catch in August, 1975, and 69.8% in July, 1974. The general trend of stations show that the largest amount of flounders were caught in July; the number was reduced greatly in October and reduced even more so in the February sampling with the numbers increasing again in August, 1975. Exceptions to these results exist at two stations. At Station A4 the largest catch was taken in October and at Station A8 the largest catch of winter flounder was taken in August (Table 8).

Scup

In general the catch of scup was largest in October and least in February. Stations A2 and A9 accounted for the largest catch in October (Table 8). Like the winter flounder, it too, comprises a commercial fishery in the area.

Table 9

List of Species Collected Fewer Than
10 Times During the Sampling Period from
July 1974-August 1975

Menidia menidia

Urophycis chuss

Paralichthys oblongus

Moracanthus hispidus

Hemitripterus americanus

Myoxocephalus aeneus

Peprilus triacanthus

Alosa pseudoharengus

Alosa aestivalis

Mustelus canis

Anchoa mitchilli

Osmerus mordax

Macrozoarces americanus

Pholis gunnellus

Ammodytes americanus

Cunner

The catch of cunners show this species to be more abundantly taken during the July, 1974 sampling, with a reduced catch in October and none being taken in February. An increased yield was observed for cunners in August, 1975. The largest catch of cunners occurred at the same station, F4, in both July and August (Table 8).

Windowpane flounder

Windowpane flounders which have been collected in large amounts in other studies on Long Island Sound (Amish & Hauer, 1974; Austin *et al.*, 1973; Hauer, 1974) were caught infrequently in the area of the New London Disposal Site. They comprised 17.3% of the mean catch in October and 7.1% in July and August. Only three windowpanes were collected during the February sampling. The largest catch per hour for an individual station occurred in July at A9 (Table 8).

Tautog

The catch of tautog was almost entirely obtained during July, 1974. Only two were collected in October, 1974 and August, 1975. None were taken during the February sampling. The largest catch was taken at A2 (Table 8). This species does comprise a sport fishery in the area.

Little skate

The little skate was obtained in July, February and August samplings; the majority were collected in August. The largest contribution of little skate was taken in August at Station A8 (Table 8).

Table 8

MEAN SPECIES ABUNDANCE
 Number of fish collected per hour
 N.S. = Not sampled

Tautoga onitis

Station	July 1974	October 1974	February 1975	August 1975
A2	36	4	0	0
A4	16	0	0	0
A8	0	0	0	4
A9	4	0	0	0
C2	4	0	0	0
C6	4	0	0	0
E9	0	N.S.	0	0
F4	12	0	0	4
F8	0	4	0	0

Raja erinacea

Station	July 1974	October 1974	February 1975	August 1975
A2	28	0	0	0
A4	4	0	0	4
A8	0	0	4	52
A9	4	0	8	24
C2	8	0	0	4
C6	8	0	12	12
E9	0	N.S.	0	8
F4	12	0	4	0
F8	0	0	0	20

Table 8 (cont.)

Pseudopleuronectes americanus

Station	July 1974	October 1974	February 1975	August 1975
A2	16	4	0	0
A4	0	176	0	8
A8	276	8	100	596
A9	152	8	48	184
C2	136	20	0	6
C6	996	4	0	252
E9	0	-	4	0
F4	308	64	64	28
F8	448	12	4	148

Stenotomus chrysops

Station	July 1974	October 1974	February 1975	August 1975
A2	64	100	0	0
A4	24	0	0	0
A8	12	20	0	36
A9	48	100	0	16
C2	60	0	0	0
C6	0	4	0	8
E9	4	-	0	0
F4	4	40	0	12
F8	16	0	0	8

Table 8 (cont.)

Tautogolabrus adspersus

Station	July 1974	October 1974	February 1975	August 1975
A2	4	0	0	0
A4	52	4	0	0
A8	0	36	0	8
A9	8	4	0	0
C2	12	60	0	36
C6	8	0	0	12
E9	0	-	0	0
F4	164	164	0	0
F8	60	0	0	4

Scophthalmus aquosus

Station	July 1974	October 1974	February 1975	August 1975
A2	0	24	0	0
A4	4	0	0	8
A8	52	40	0	44
A9	108	8	8	20
C2	4	0	4	0
C6	4	4	0	16
E9	4	-	0	0
F4	20	20	0	8
F8	36	80	0	16

Table 8 (cont.)

Prionotus carolinus

Station	July 1974	October 1974	February 1975	August 1975
A2	0	0	0	0
A4	0	4	0	0
A8	0	0	0	32
A9	0	0	0	4
C2	0	0	0	0
C6	4	0	0	4
E9	0	-	0	0
F4	12	4	0	8
F8	16	8	0	0

Paralichthys dentatus

Station	July 1974	October 1974	February 1975	August 1975
A2	0	0	0	0
A4	0	0	0	0
A8	0	0	0	0
A9	0	0	0	4
C2	0	0	0	0
C6	4	0	0	0
E9	0	-	0	0
F4	12	0	0	0
F8	16	0	0	0

Table 8 (cont.)

Myoxocephalus octodecemspinosus

Station	July 1974	October 1974	February 1975	August 1975
A2	0	0	0	0
A4	0	0	0	0
A8	0	0	16	0
A9	0	0	8	0
C2	0	0	0	0
C6	0	0	0	0
E9	0	-	4	0
F4	0	0	12	0
F8	0	0	32	0

Northern searobin, summer flounder and longhorn sculpin

The three remaining species were all caught very infrequently. The northern searobin and summer flounder were caught almost exclusively during the summer samplings (Table 8). The longhorn sculpin was only caught during the February sampling (Table 8).

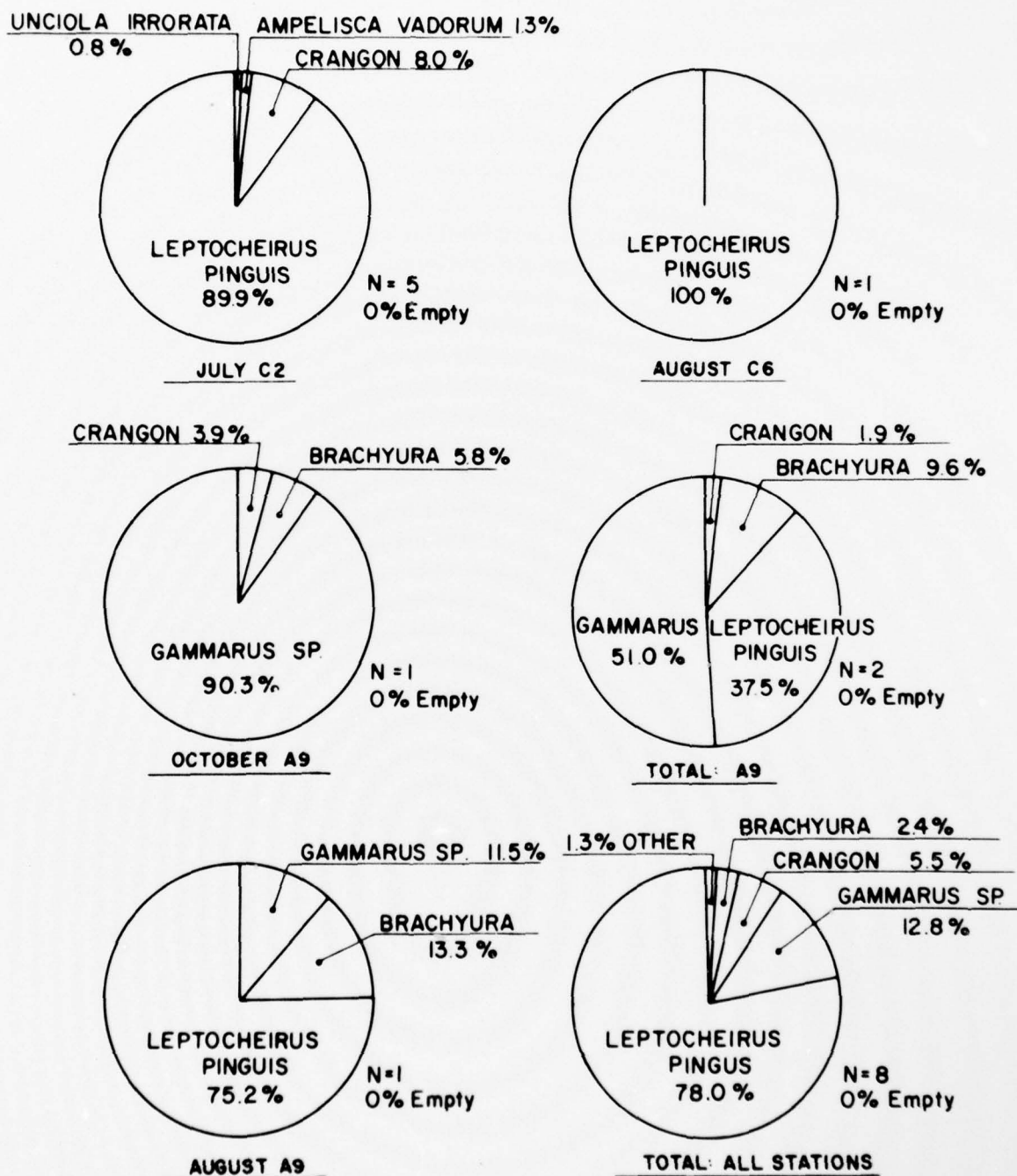
2. Stomach Content Analysis

During the sampling period, the analysis of stomach contents was performed on seven species of demersal fish. These fish include: northern searobin, *Prionotus carolinus*; winter flounder, *Pseudopleuronectes americanus*; little skate, *Raja erinacea*; windowpane flounder, *Scophthalmus aquosus*; scup, *Stenotomus chrysops*; tautog, *Tautoga onitis*, and cunner, *Tautoglabrus adspersus*. The fish collected at each sampling site were preserved in a buffered 10% formalin solution and returned to the laboratory for further investigation. Of the nine stations sampled, three representative stations were chosen to be used in this analysis. The three stations are: Station A9, directly to the east of the disposal site; Station C6, which is found within the boundaries of the disposal site, and Station C2, located to the west of the disposal site.

Northern searobin

Prionotus carolinus (see Figure 3) was caught at each station, and the predominant food item was found to be the amphipod *Leptocheirus pinguis*. The 1.3% noted as "other" in the total for all stations included the amphipods,

Figure 3
STOMACH CONTENTS OF
Prionotus carolinus



Ampelisca vadorum and *Unciola irrorata*, and a small amount of unidentified digested material.

Winter flounder

Pseudopleuronectes americanus (see Figures 4-6) was found to be the fish with the most diverse diet. This fish was also caught in the greatest numbers, usually with the quota of fifteen fish being collected per station per sampling period. The most common food item was found to be the amphipod, *Gammarus* sp., with a variety of polychaetes and amphipods comprising most of the remainder. The greatest percentage of empty stomachs was found during the February sampling. This should be expected, since that sampling took place during the flounders' reproductive season, and it has been reported that winter flounder refrain from eating while gravid. The 10.3% described as "other" in the total for all stations included: *Unciola irrorata*, *Crangon septemspinosus*, *Lepidonotus squamatus*, *Caprella* sp., *Musculus niger*, *Hypaniola grayi*, *Glycera americana*, *Pectinaria gouldii*, *Ampharete acutifrons*, Chordata, Nemertean, *Axiiothella* sp., *Mulinia lateralis*, *Neomysis americana*, *Corophium* sp., *Photis* sp., Echinodermata, Nematoda, crab larvae, sponge, various unidentifiable bivalves, and bits of wood and sand.

Little skate

Raja erinacea (see Figures 7-8) were collected at all stations during the sampling period, and were found to have a fairly diverse diet. The dominant

Figure 4

STOMACH CONTENTS OF
Pseudopleuronectes americanus

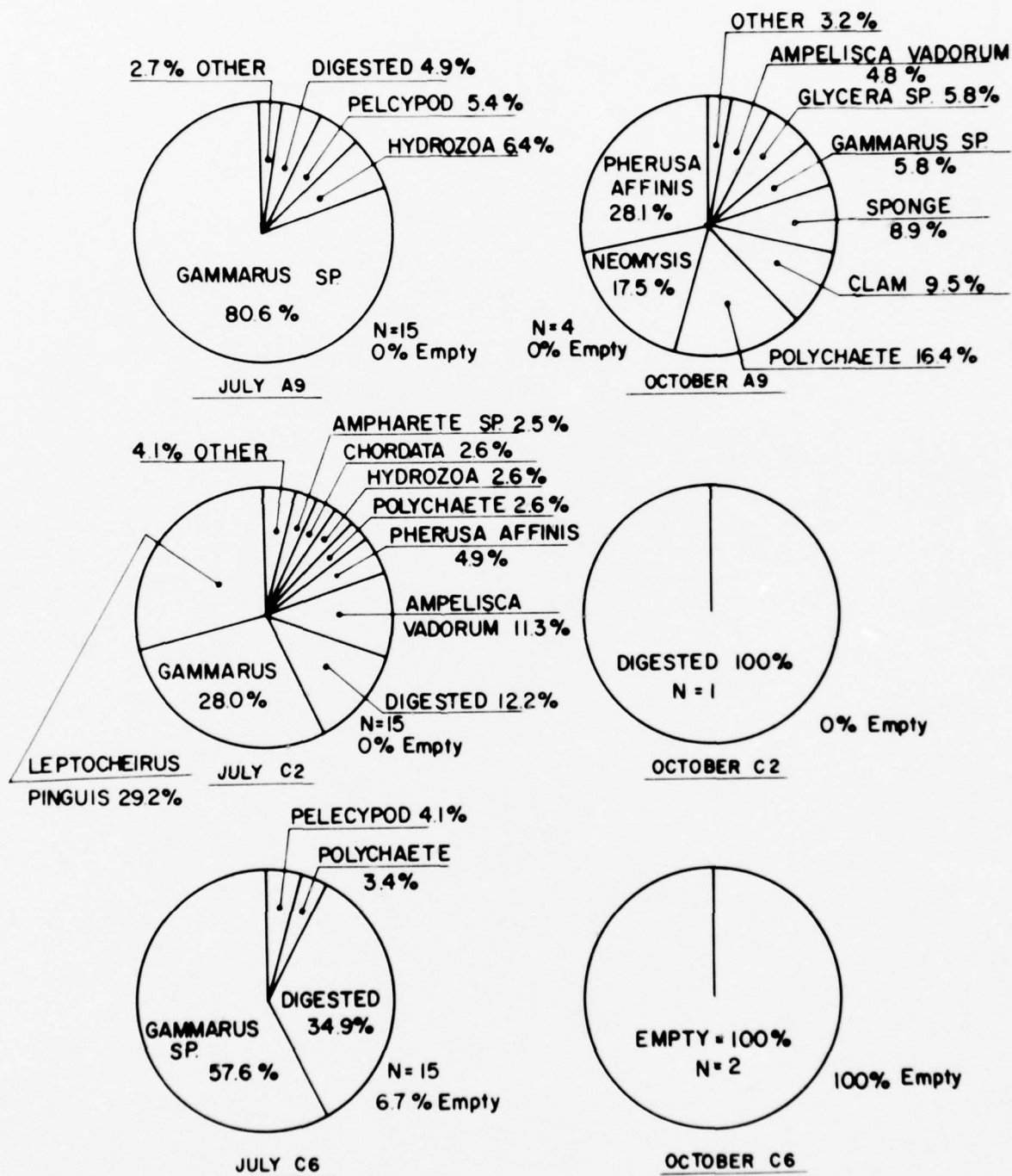


Figure 5

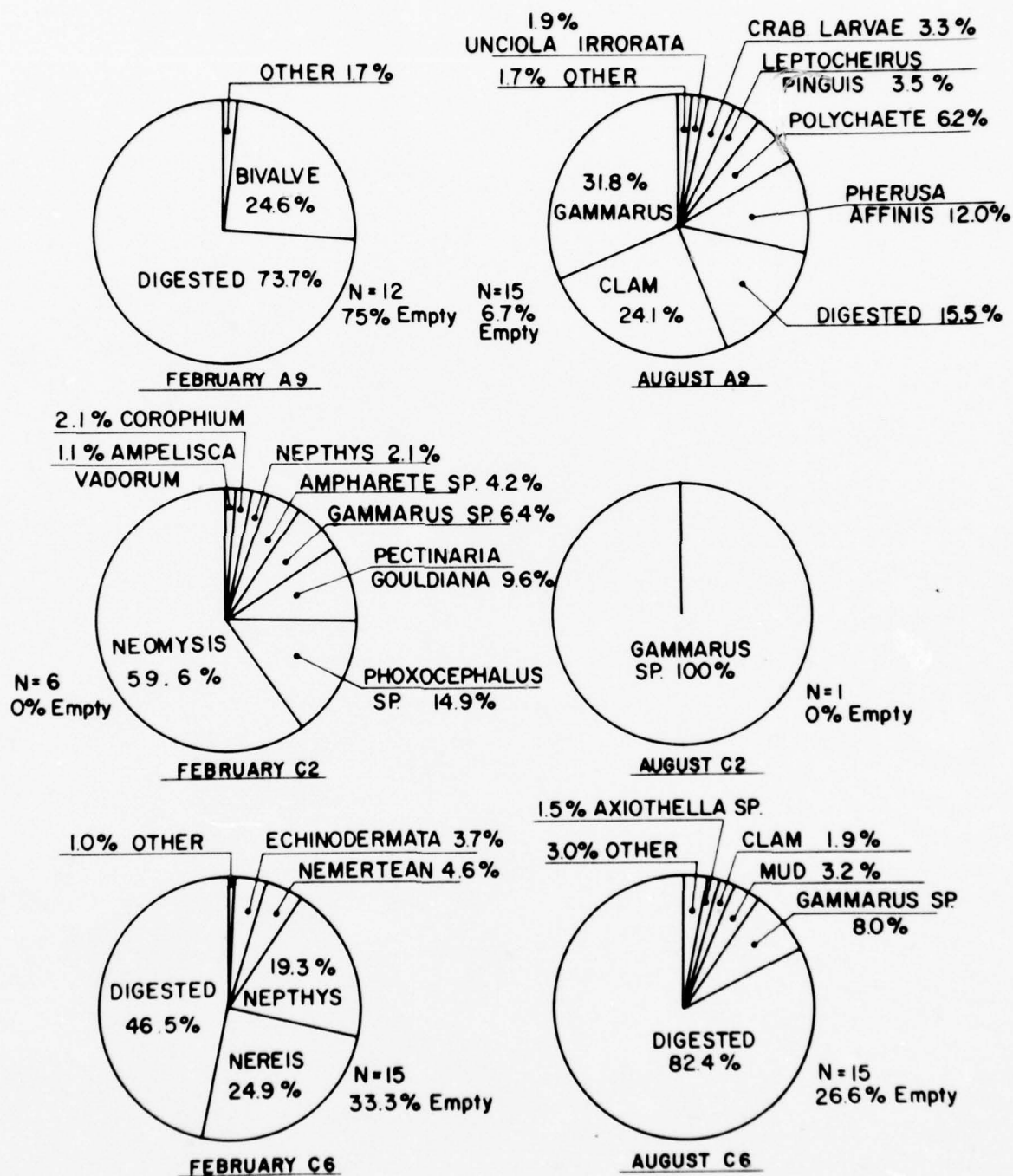
Pseudopleuronectes americanus

Figure 6

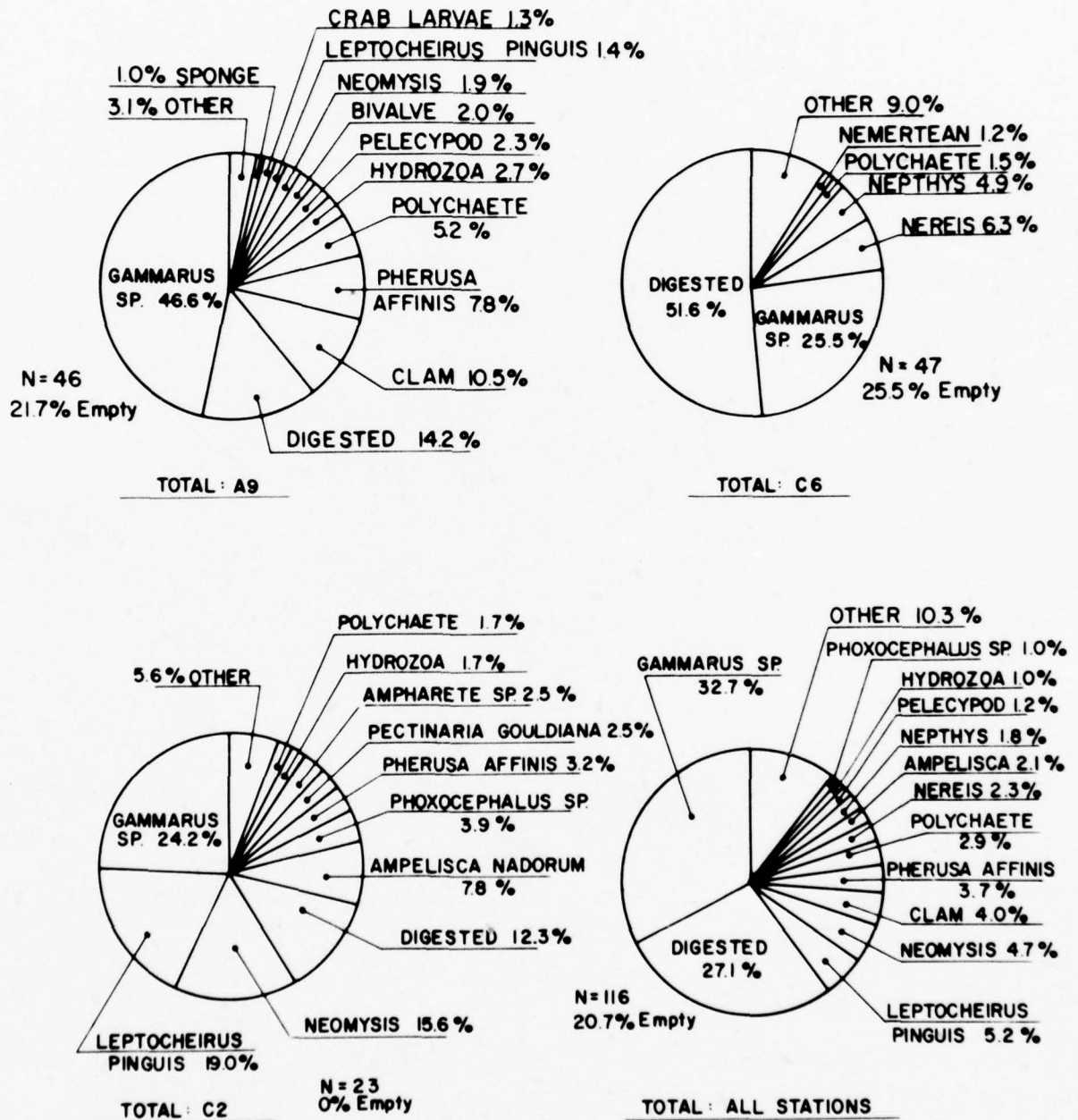
Pseudopleuronectes americanus

Figure 7
STOMACH CONTENTS OF
Raja erinacea

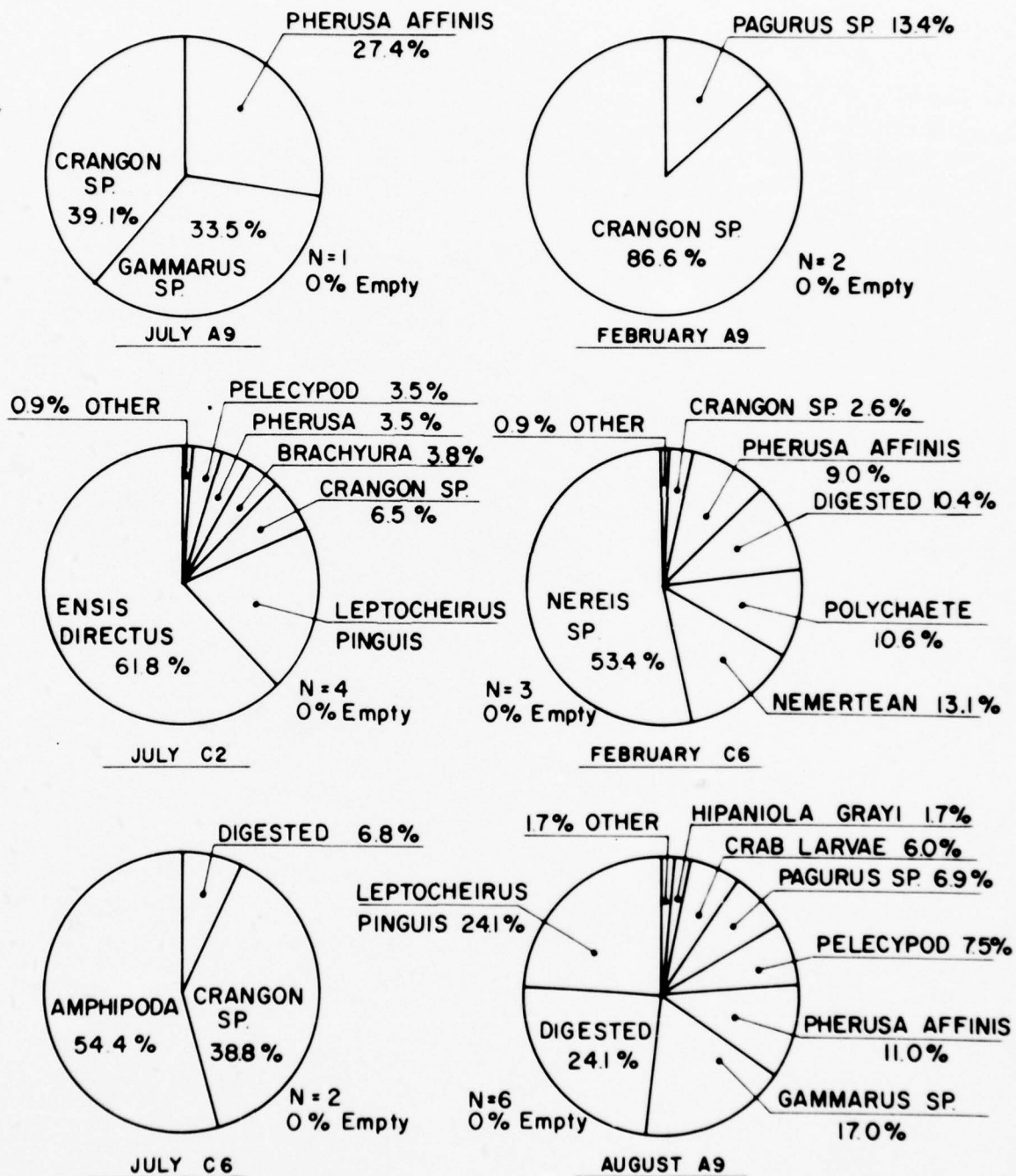
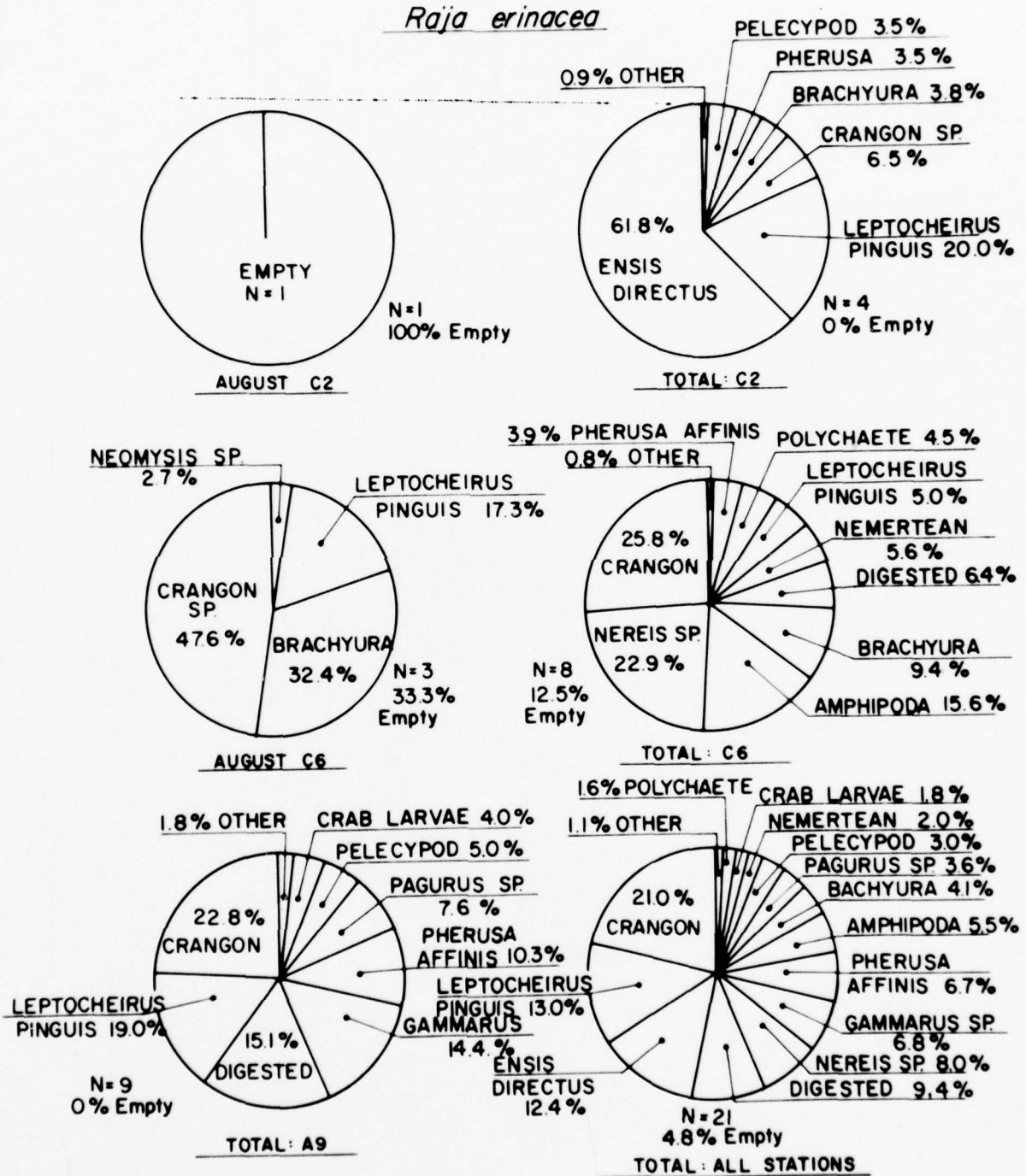


Figure 8



food item was the decapod, *Crangon septemspinosus*, with other amphipoda and polychaeta following close behind. The 1.1% noted as "other" in the total for all stations was composed of *Hypanicola grayi*, *Neomysis americana*, *Unciola irrorata*, fish and shell fragments. The skate was found to have a relatively low percentage of empty stomachs, when compared with other species sampled.

Windowpane flounder

Scophthalmus aquosus (see Figures 9-11) were obtained from each sampling site, and upon examination, proved to have a somewhat limited diet. The predominant food item was the mysid, *Neomysis americana*, with *Crangon septemspinosus* following close behind. At Station C6, however, the dominant item was *Loligo* sp. with *Neomysis americana* second. The 1.9% listed as "other" in the total for all stations was comprised of crab larvae, *Leptocheirus pinguis* and bits of rock.

Scup

Stenotomus chrysops (see Figures 12-13) were collected at all stations, and during all sampling periods, except for the February sampling when none were caught. They were found to have a diverse diet, although not to such a degree as the winter flounder. In comparison to other species sampled, fish at each station had a different major food item. The 2.7% labeled as "other" under the total for all stations consisted of *Caprella* sp., unidentified amphipoda, *Ampharete acutifrons*, unidentified polychaeta, *Photis* sp., Pelecypoda, Crustacea and copepods.

Figure 9
STOMACH CONTENTS OF
Scophthalmus aquosus

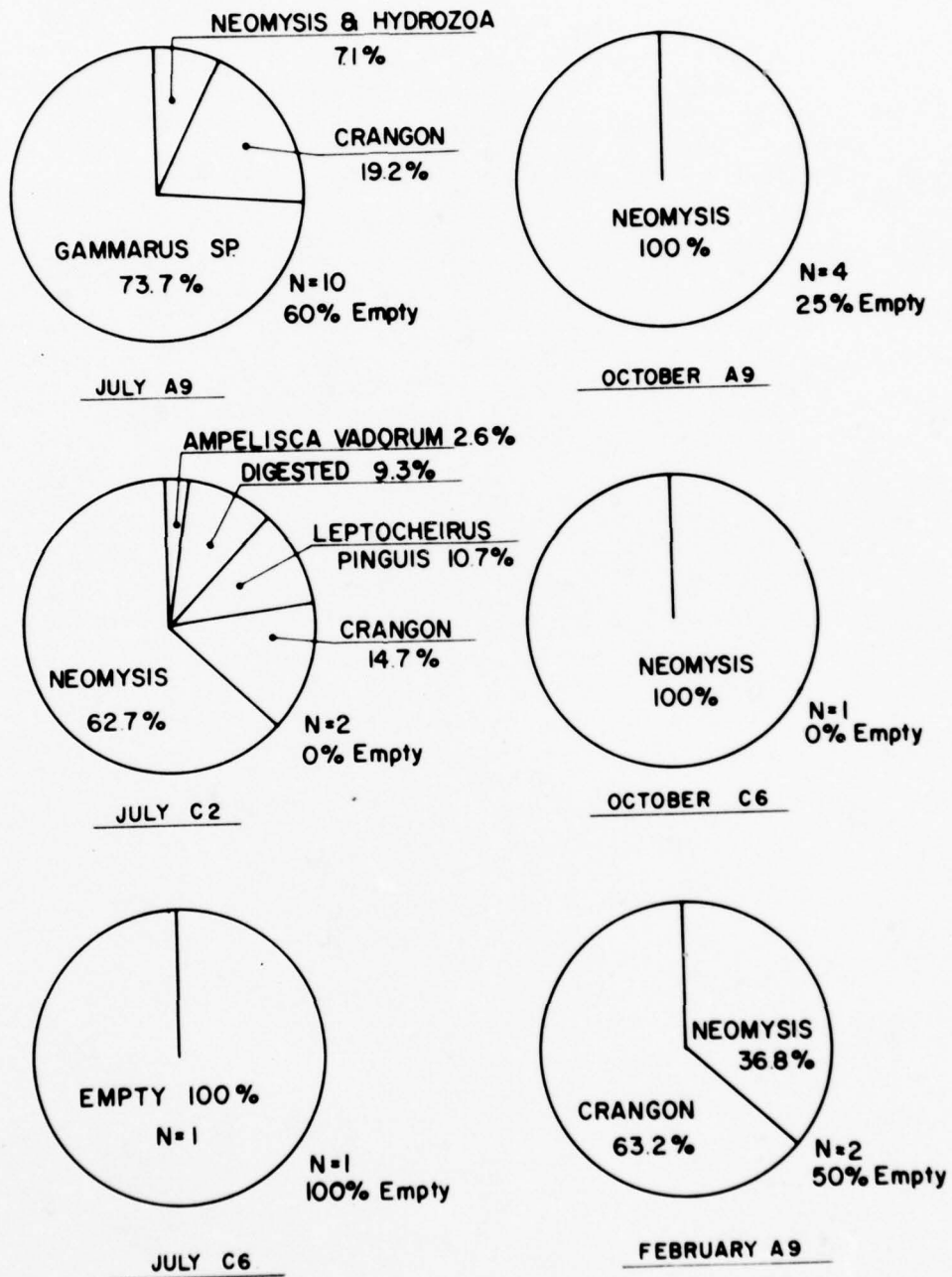


Figure 10

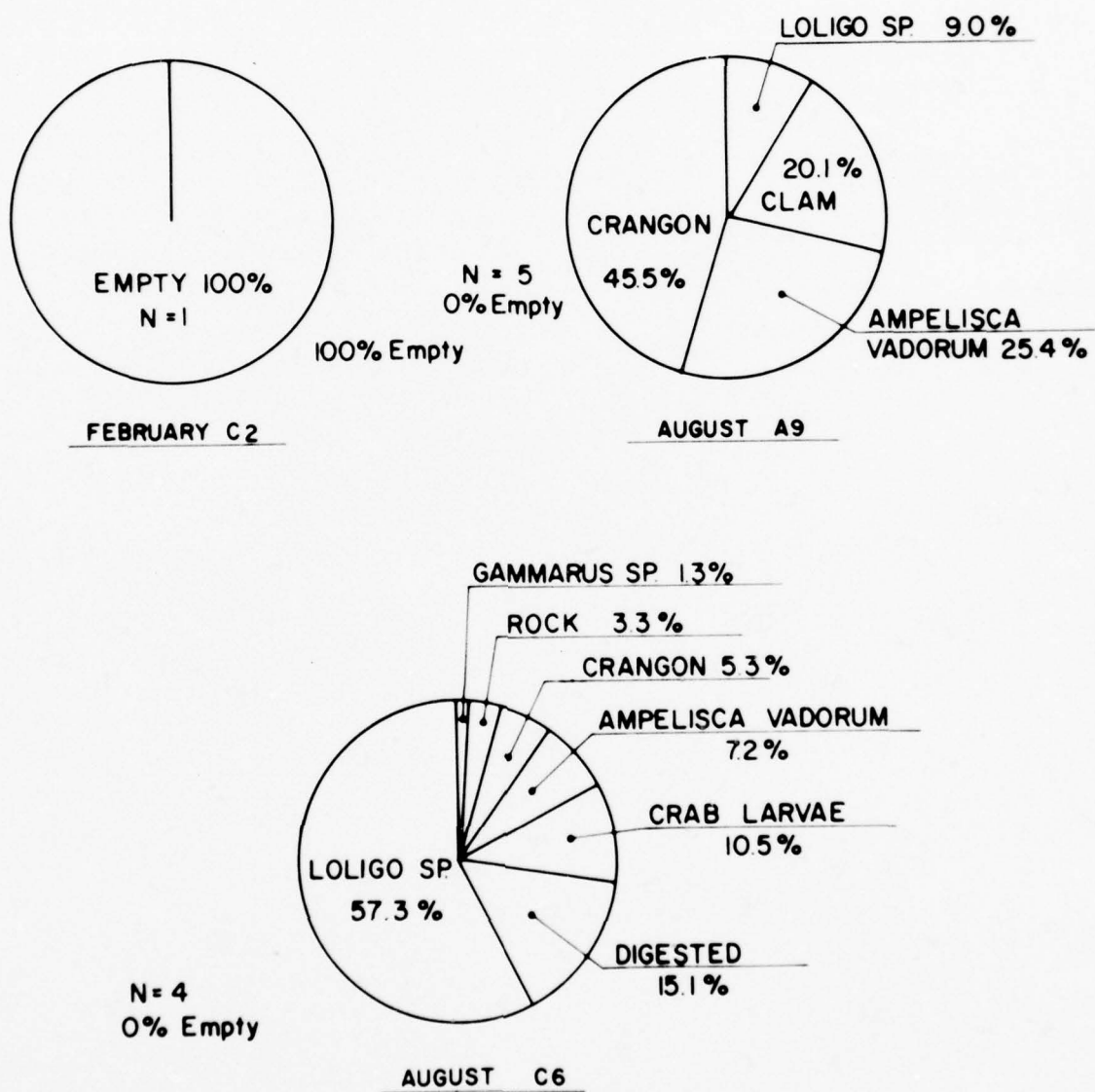
Scophthalmus aquosus

Figure II

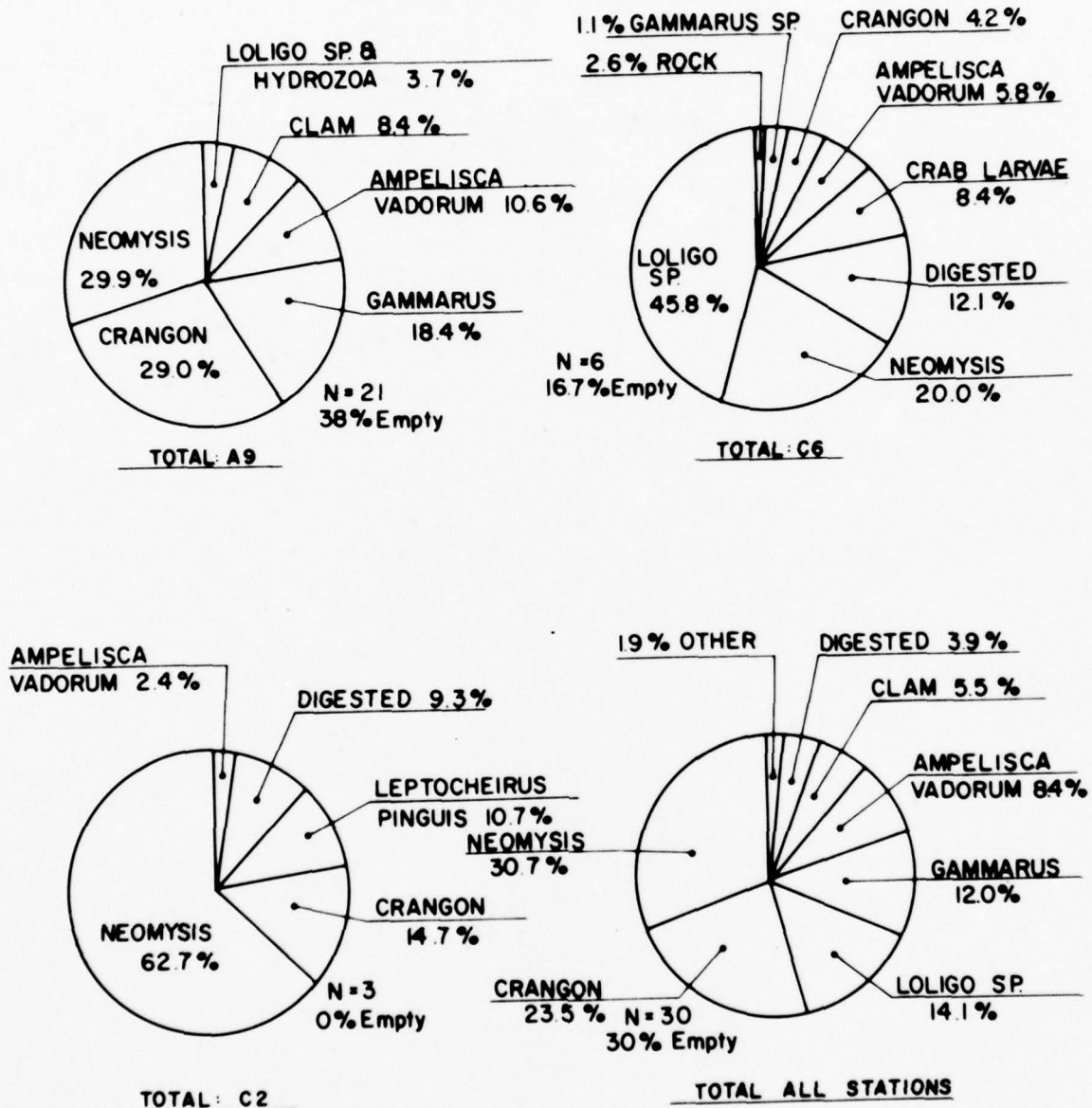
Scopthalmus aquosus

Figure 12
STOMACH CONTENTS OF
Stenotomus chrysops

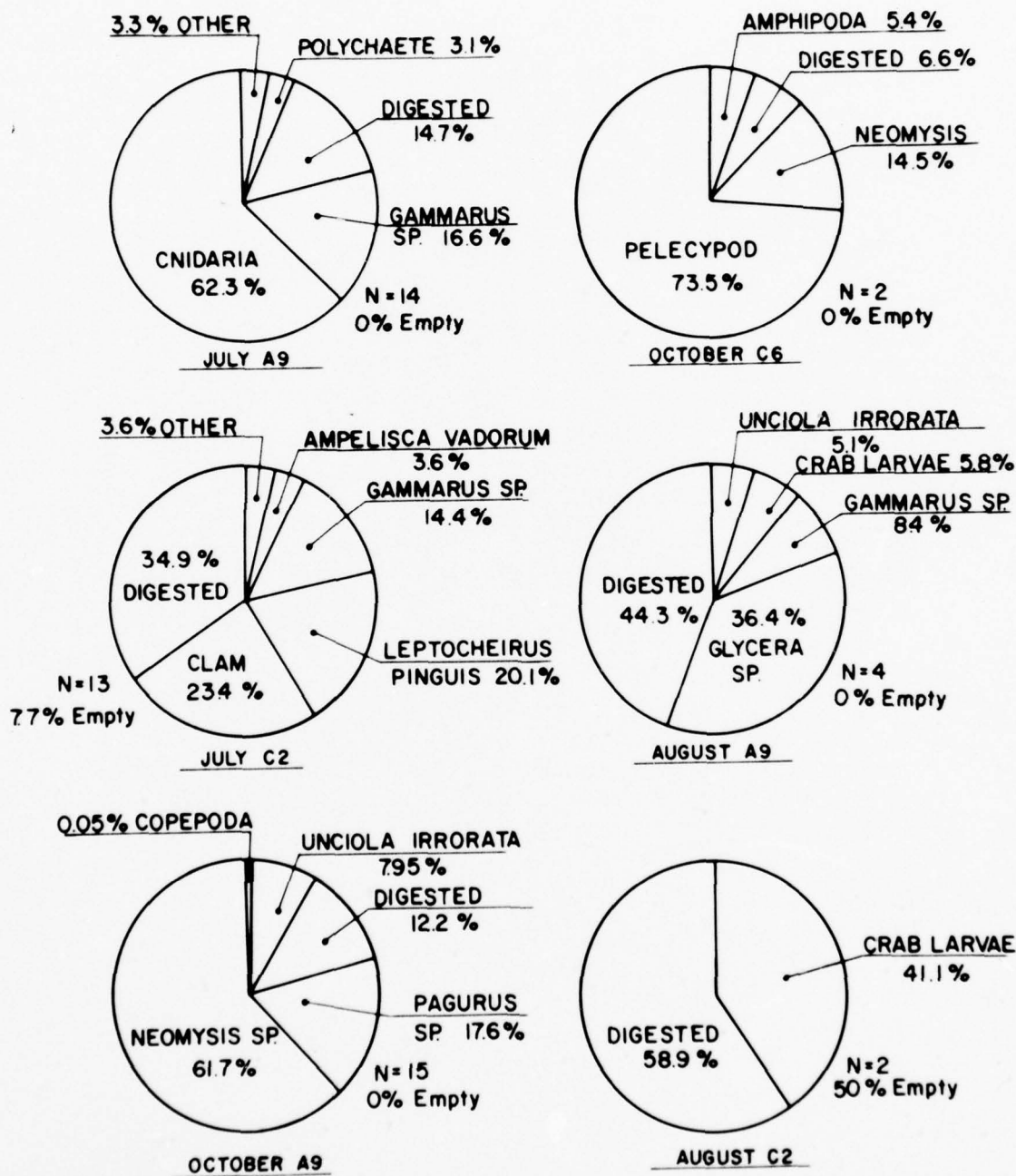
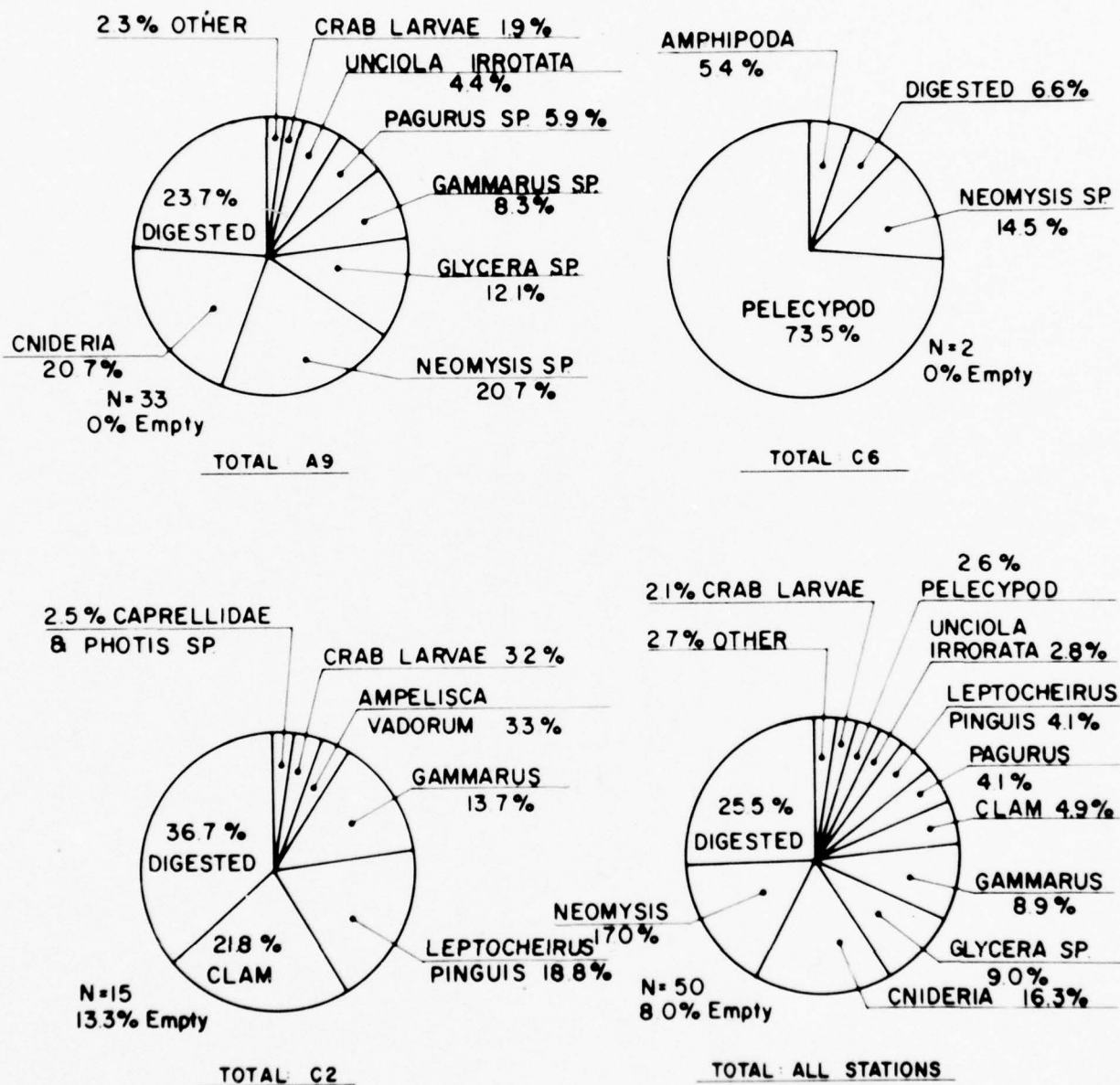


Figure 13

Stenotomus chrysops

Tautog

Tautoga onitis (see Figure 14) were only collected at Stations A9 and C2. Their diet was comprised almost entirely of *Pagurus* sp. and *Cancer* sp.

Cunner

Tautogolabrus adspersus (see Figures 15-16) were collected at all stations and during each sampling period, except for the February sampling, during which none were collected. The cunner may be described as an opportunistic feeder. Upon examination, their diet proved to be relatively diverse, with the fish at each station having a different major food item. The 1.6% noted as "other" under the total for all stations consisted of *Crangon septemspinosus*, *Caprella* sp., *Nassarius* sp., *Musculus niger*, *Brachyura*, unidentifiable fish and algae.

3. Age and Size Range

The ages and sizes of fish collected at the New London Disposal Site were determined for four species and are presented in Table 10 and Figure 18. These species are: northern searobin, *Prionotus carolinus*; winter flounder, *Pseudopleuronectes americanus*; windowpane flounder, *Scophthalmus aquosus*, and scup, *Stenotomus chrysops*. Ages were determined by counting annuli on otoliths and/or scales. All lengths are recorded as standard length. Due to the preserving process, the age determination of such fish as tautog, *Tautoga onitis* and cunner, *Tautogolabrus adspersus* often could only be approximated. The results were inconsistent and thus the data are not reported. The fish used in this determination were selected from three representative stations (Stations A9, C2, and C6; see Figure 1).

Figure 14
STOMACH CONTENTS OF
Tautoga onitis

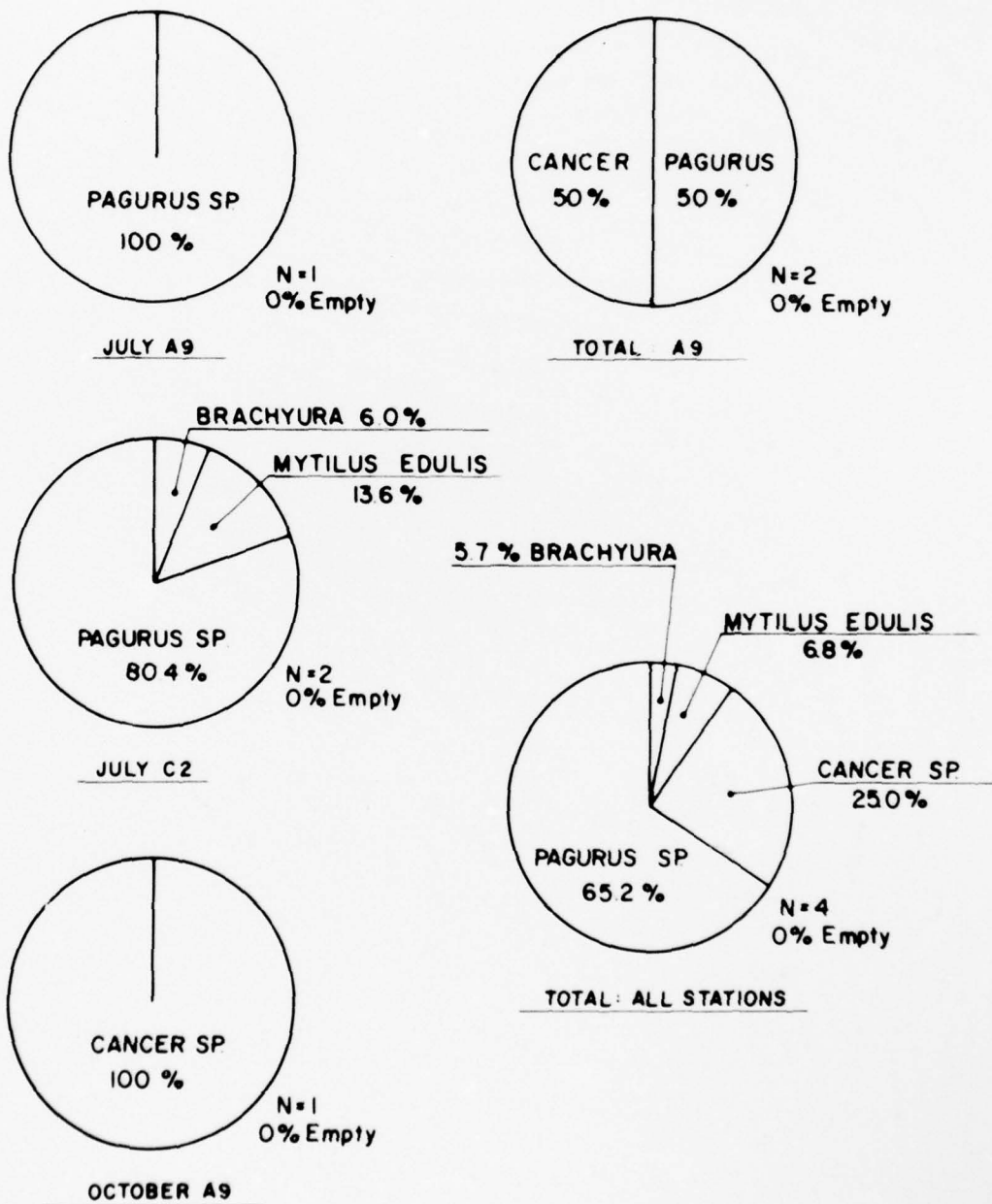


Figure 15
STOMACH CONTENTS OF
Tautoglabrus adspersus

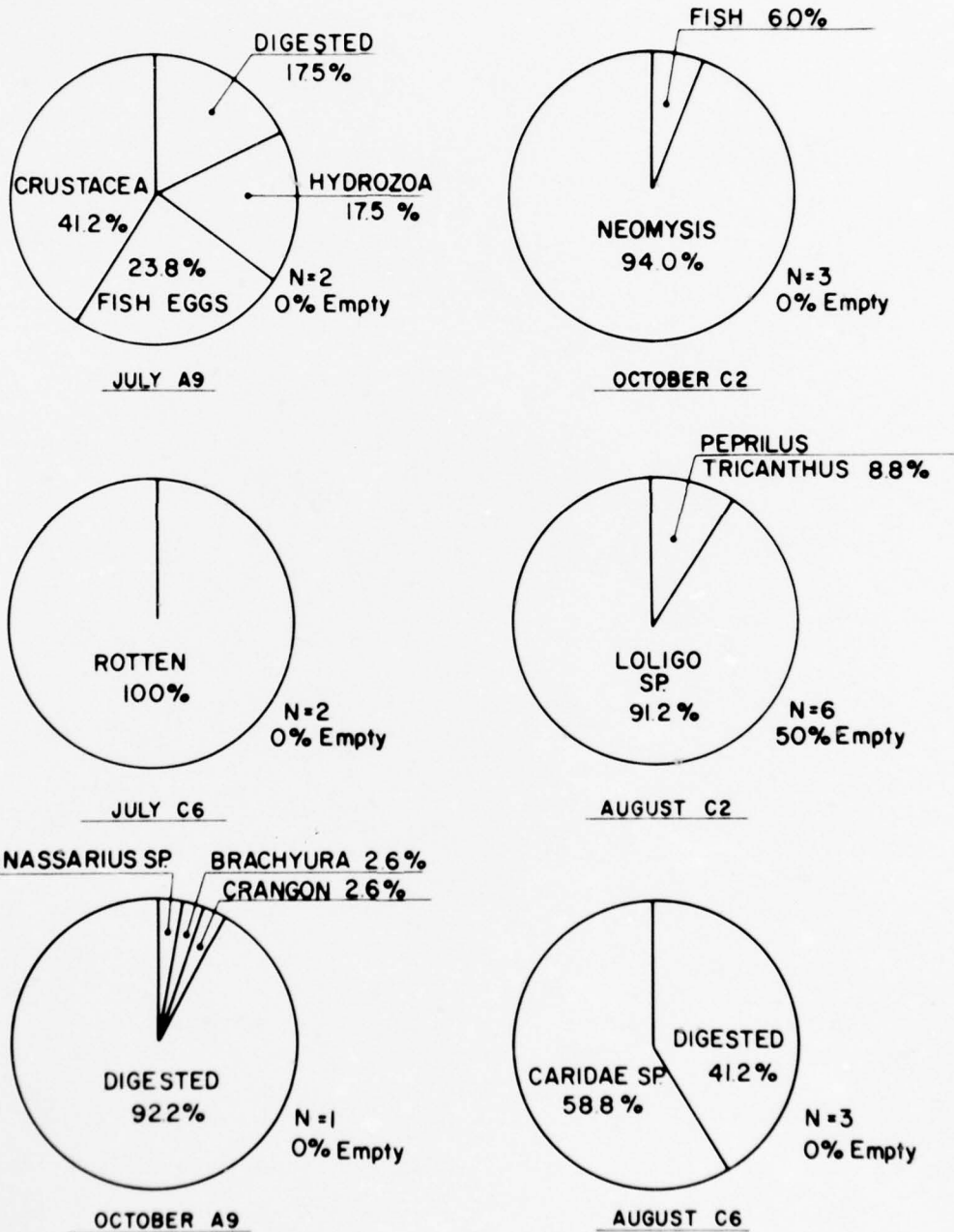
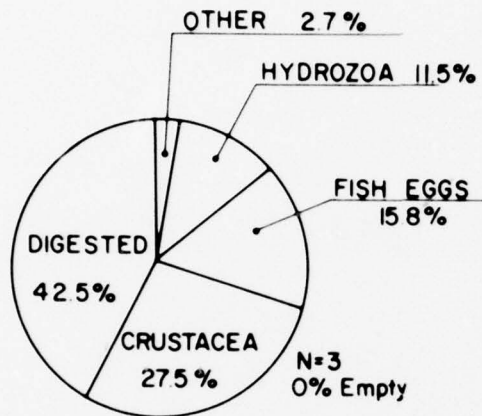
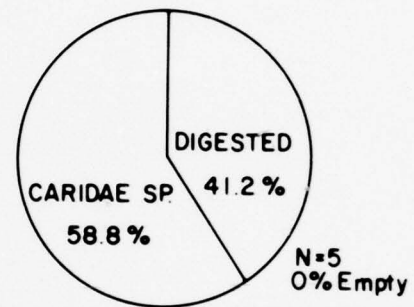


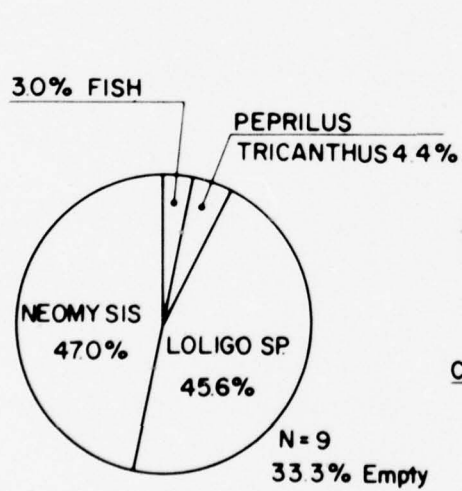
Figure 16

Tautoglabrus adspersus

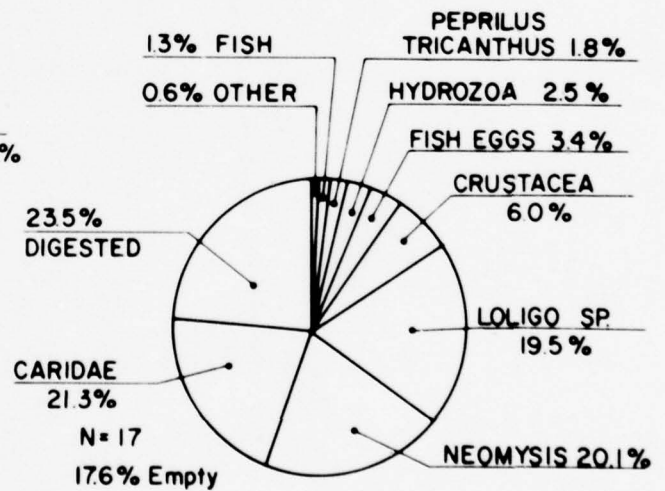
TOTAL: A9



TOTAL: C6



TOTAL: C2



TOTAL: ALL STATIONS

Figure 17

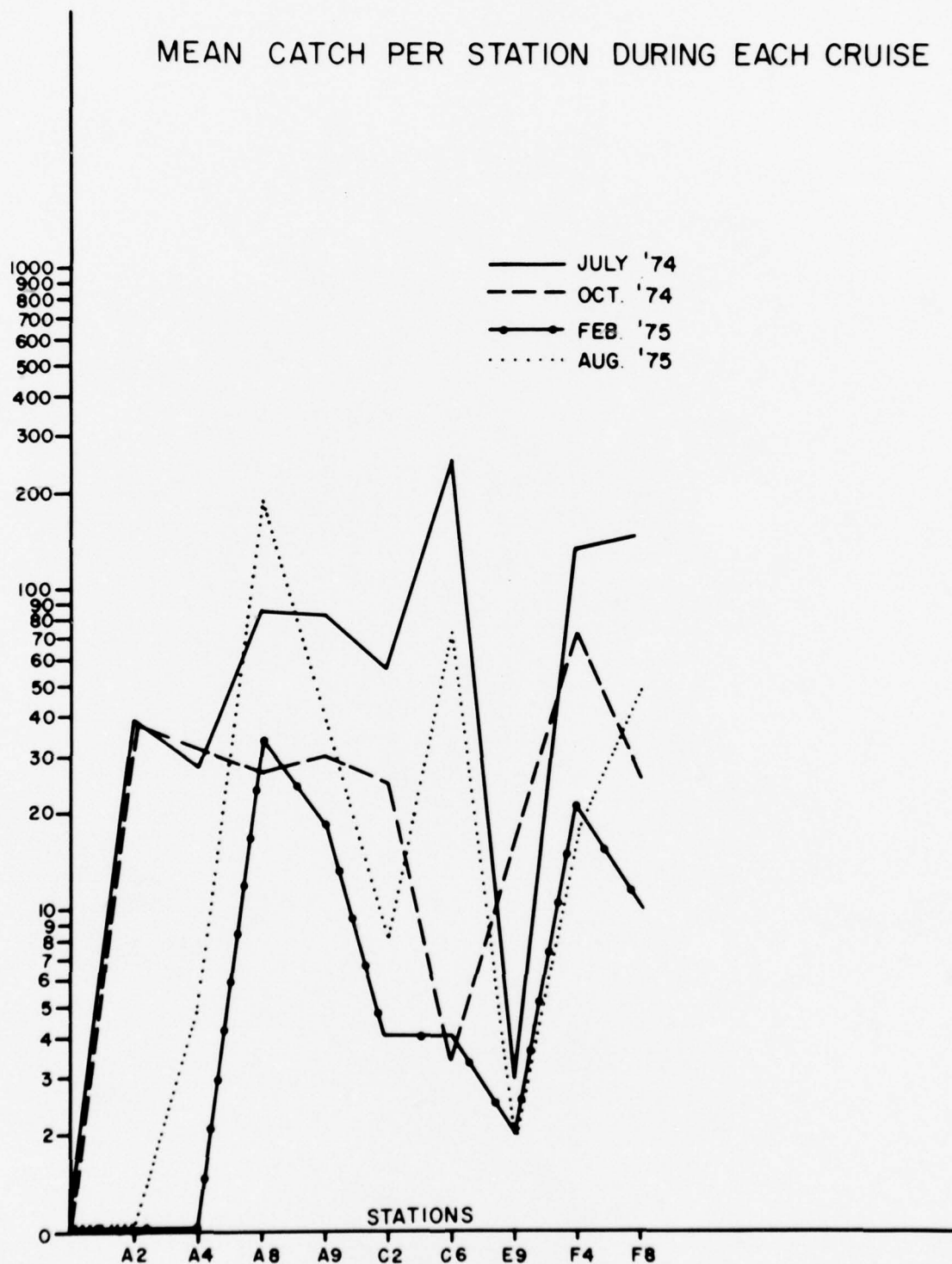


Table 10

AGE & MEAN STANDARD LENGTH RANGE RELATIONSHIP FOR FOUR
SPECIES COLLECTED AT THE NEW LONDON DISPOSAL SITE

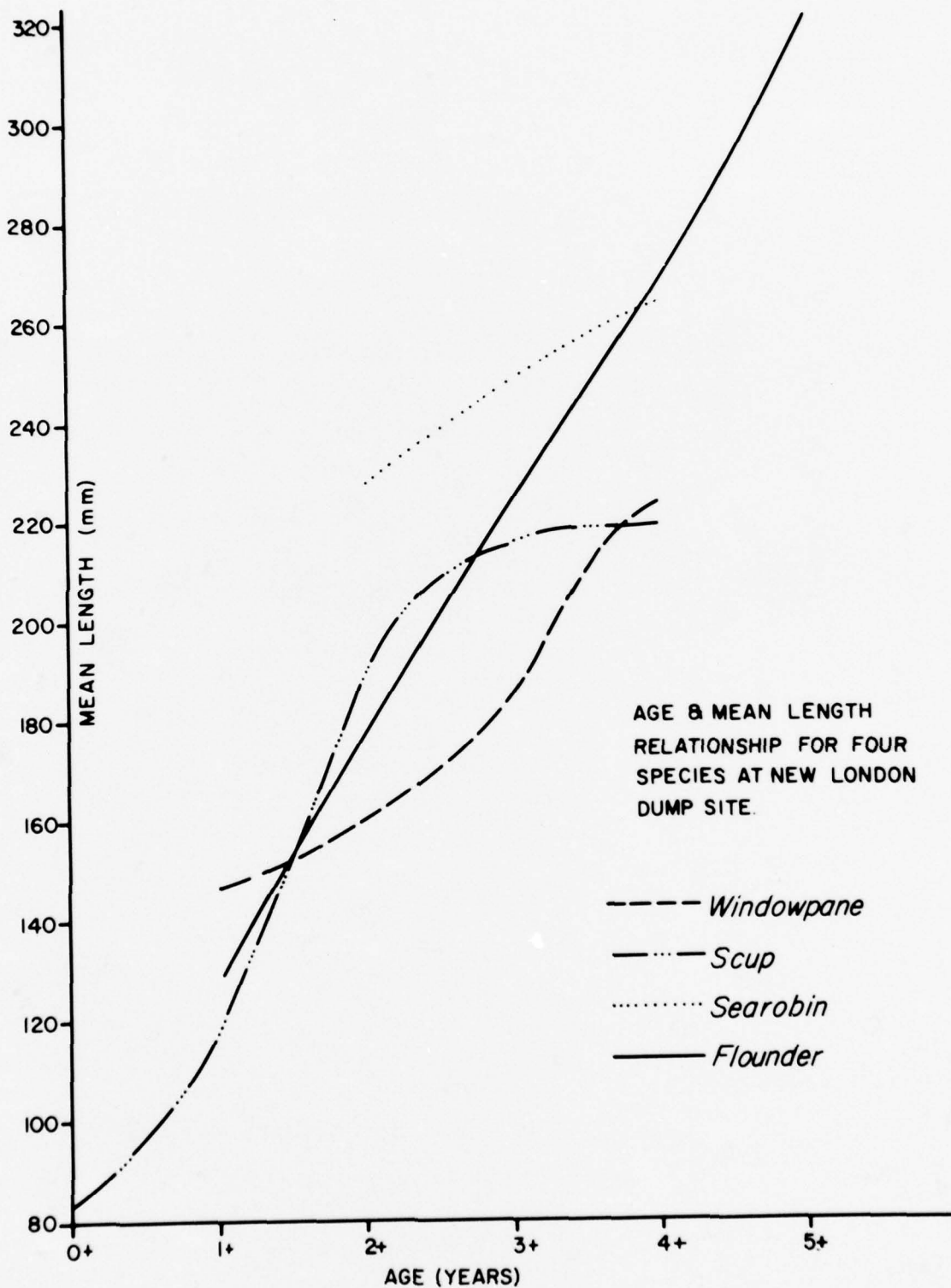
<u>P. americanus</u>	<u>N</u>	<u>%</u>	<u>\bar{x} (mm)</u>	<u>Range (mm)</u>
0 +	0	0	-	- - -
1 +	48	38.1	128	98 - 172
2 +	44	34.9	179	164 - 210
3 +	25	19.8	225	195 - 264
4 +	6	4.8	269	253 - 283
5 +	3	2.4	323	314 - 332

<u>S. aquosus</u>				
0 +	0	0	-	- - -
1 +	1	3.2	147	147
2 +	16	51.6	161	142 - 195
3 +	12	38.7	185	165 - 203
4 +	2	6.5	224	202 - 246

<u>S. chrysops</u>				
0 +	13	24.1	83	77 - 90
1 +	31	57.4	118	90 - 152
2 +	6	11.1	186	164 - 212
3 +	2	3.7	216	198 - 235
4 +	2	3.7	220	220 - 221

<u>P. carolinus</u>				
0 +	0	0	-	- - -
1 +	0	0	-	- - -
2 +	5	62.5	228	222 - 234
3 +	1	12.5	250	250
4 +	2	25.0	265	260 - 270

Figure 18



SUMMARY

The demersal fish populations at the New London Disposal Site were sampled four times. The first sample, which took place before disposal operations began, determined baseline conditions at the site. The second and third sampling periods were scheduled to coincide with disposal. The fourth sample was taken after the dredged material disposal was completed.

Of the 1,577 fish collected over the entire four sampling periods, 56% were obtained before disposal began. The landings of fish dropped sharply when disposal operations began and never recovered to the pre-disposal baseline level. This variation, however, may be due to seasonal abundance rather than the effects of the dredge spoil. The most abundant species collected throughout the study was winter flounder.

The stomach contents of seven species of demersal foraging fish were examined. Comparisons were observed between the pre-disposal and post-disposal diets in Figures 3-16. All of the species collected, with the exception of windowpane flounder and tautog, had a varied diet and may be described as opportunistic feeders. The windowpane flounder and tautog were found to be very selective in their food habits.

Ages and size ranges were determined for four species of fish collected at the disposal site.

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H. SCUBA survey and underwater photography of the New London disposal site,
May 1974 to June 1976.

Lance L. Stewart
Robert DeGoursey

Introduction & Methods

Underwater observations by University of Connecticut divers were conducted to note and record in-situ biological and physical processes of the dredge-disposal operation that would be difficult to detect by shipboard experiment. Initial efforts were directed to assess the types and range of benthic habitat found within the spoil area, and the impact of disposal on commercially important species with emphasis on the lobster.

Dives were made at selected stations (the four corners and one central location, Figure 1) on the designated disposal site to determine bottom characteristics and typical macrobenthos within the area. Data recorded on each dive included: depth, tidal stage, bottom visibility, estimate of current velocity and direction, bottom type, and predominant benthic species. Compass transects were followed on each dive with all notable observations recorded by 35 mm still or 16 mm motion photography. Average bottom time for transects was approximately 25 minutes, courses followed were perpendicular or oblique to bottom current, and distance ranged from 100-200 meters.

Throughout the period of spoil disposal, dives were made directly on the release area (N.L. Buoy) to note features of bottom coverage, relative turbidity, and biological colonization of the dredged material. Photographs were taken along several transects, out to the spoil periphery, to provide visual impression of the conditions and composition of dredged deposits. Spot dives were made at disposal site corners (approximately .5 n. miles from New London Buoy) to check for evidence of spoil dispersal.

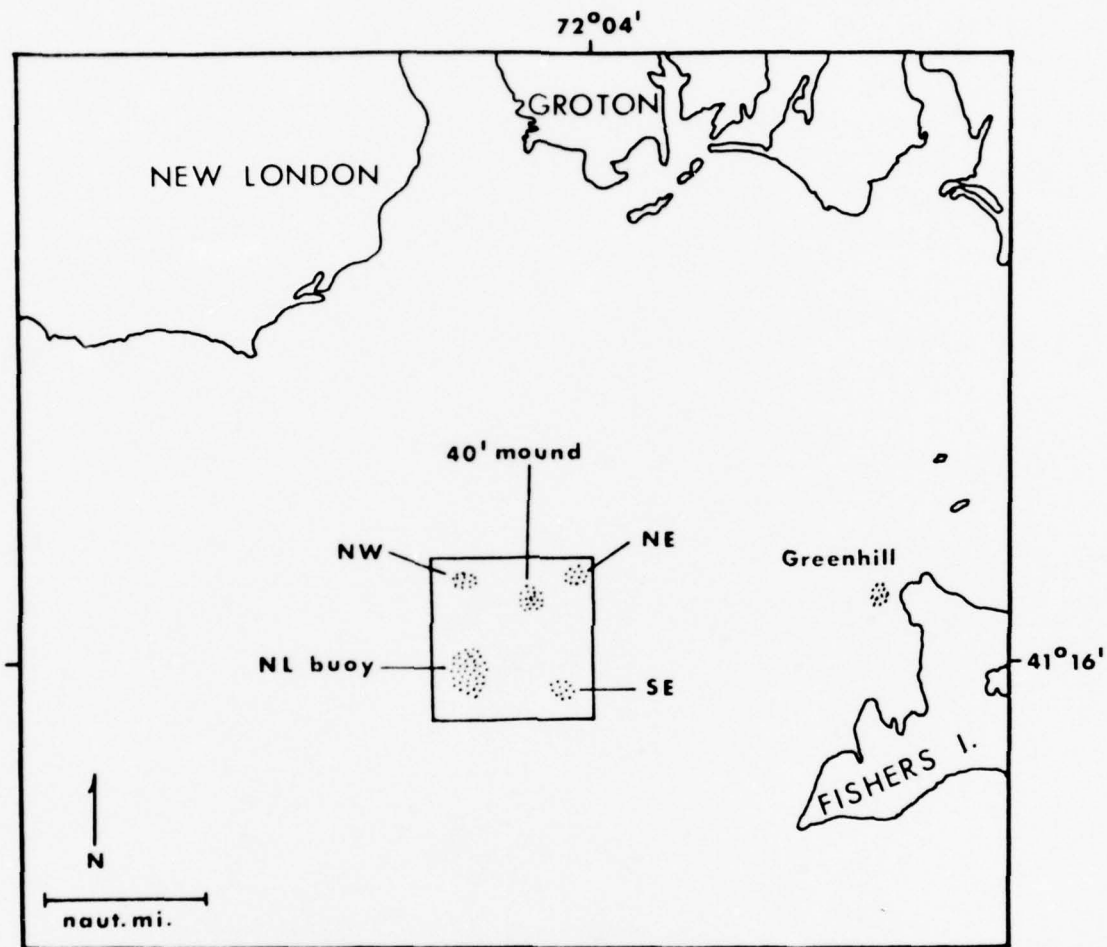


Figure 1. Locations of dive survey sites on the New London disposal area, May-October 1974.

Underwater 35 mm photographs with descriptive text were provided for the second quarterly report (No. 49) and 16 mm color footage was edited and made available for viewing. Copies of selected 35 mm slides and 16 mm footage were duplicated for use by personnel at: The National Marine Fisheries Service Laboratory, Sandy Hook, New Jersey; and the U.S. Navy.

Results

Table I lists the date, location, and activity for all University of Connecticut dive surveys to date. A series of sample photographs (Figures 2-19) illustrates pre-disposal bottom conditions, lobster habitat, characteristics of recently dumped spoil and evidence of biological colonization. Complete film records (35 mm and 16 mm) have been indexed and are on file at the Marine Advisory Service Office, bldg. 24, Avery Point.

Investigation of general benthic habitat throughout the disposal area did not reveal bottom type supporting high populations of either lobster, Homarus americanus, or crab, Cancer (sp.). Lobster burrows were scarce, located beneath debris or scattered boulders (Figures 7 & 9), or constructed directly into mud substrate (Figure 8). Counts of individual lobsters observed on transect dives ranged from three to ten with highest numbers recorded in the north central mound vicinity. Productive rock reef areas adjacent to the disposal site would yield counts of 20-30 lobsters per dive. Newly dumped spoil provided a relief to the flat featureless bottom that appeared mildly attractive to lobsters and crabs. Throughout the disposal period, freshly excavated burrows were noted along the periphery and in clay mounds on the spoil pile. However, it is believed continued dumping activity and the gradual settling of the pile did not allow the formation of an increasing number of burrow locations.

Table I. Date and location of dives on the New London Disposal Site for Benthic survey and underwater photography, May, 1974-June, 1976.

Date	Location	Benthic Survey	Underwater Photography	
			35 mm	16 mm
<u>1974</u>				
21 May	Northeast corner	x	x	
13 June	Southeast corner	x	x	x
26 June	Northwest corner	x	x	x
	N.L. Buoy	x	x	
8 July	North central (40' mound)	x		x
2 August	Northeast corner	x		x
9 September	Northeast corner			
	Southeast corner			
	Northwest corner	x	x	
16 September	N.L. Buoy (north transect)	x	x	
14 October	N.L. Buoy (south transect)	x	x	
<u>1975</u>				
7 March	N.L. Buoy (southeast transect)	x	x	
28 May	N.L. Buoy (south transect)	x	x	x
23 July	300 yds. East of N.L. Buoy (S.W.)	x	x	
<u>1976</u>				
10 May	N.L. Buoy (ENE transect)	x	x	
16 June	300 yds. NE of N.L. Buoy		x	
	N.L. Buoy (east transect)	x		
	.5 miles SE of N.L. Buoy	x		

Underwater Photographs at Selected Stations at the New London Dump Site

General Bottom Types:

- Figure 2. 5/21/74; N.E. Corner; Depth 52'; Horizontal field view of the photo (HFVP), 10". Natural erosion zone around masses of amphipod tubes with an exposed tube of Diopatra cuprea (upper center). General sediment condition compact and apparently consolidated by the infauna.

- Figure 3. 5/21/74; N.E. Corner; Depth 52'; HFVP 8". Sponge Desmacidon sp. attached to shell fragments in dense concentration of amphipod tubes. Natural settling of suspended matter resulted in a fine silt veneer over relatively compact bottom sediment.

- Figure 4. 5/21/74; N.E. Corner; Depth 52'; HFVP 18". The winter flounder, Pseudopleuronectes americanus, was abundant at all sites and frequently associated with amphipod communities.

- Figure 5. 6/26/74; N.L. Buoy; Depth 80', HFVP 18". Evidence of previous dumping of dredge spoils of estuarine origin indicated by the presence of oyster and bay scallop shells. Encrustation of Halichondria growth and squid egg clusters were often encountered.

- Figure 6. 9/9/74; S.E. Corner; Depth 85'; HFVP 5". Bloodstar, Henricia sanguinolenta, was a major component of the epibenthic fauna at all stations. The heavy shell fragments and sand-mud substrate were similar to the substrate observed at N.L. Buoy site.

- Figure 7. 6/26/74; N. L. Buoy; Depth 80'; HFVP 36". Typical lobster burrow in substrate, showing excavated materials, range of cobble-boulder sizes, and associated fauna.

- Figure 8. 6/26/74; N.L. Buoy; Depth 75'; HFVP 30". Lobster burrow excavated in cohesive mud-silt substrate rich in amphipod tubes.

- Figure 9. 5/21/74; N.E. Corner; Depth 50'; HFVP 20". Lobster burrow constructed under waterlogged wood from previous dumping operations. Wooden and steel beams, brass pipes and iron bulkheads were often sighted and provided attachment surfaces or shelter for benthic organisms.

Dredge Spoil Characteristics:

- Figure 10. 9/16/74; N.L. Buoy (N); Depth 70'; HFVP 6". Finfish, crustaceans and fouling organisms were seen to colonize newly deposited dredge spoils. Photo shows a Mercenaria shell embedded in clay with a newly settled anemone, Diadumene leucolena (arrow).
- Figure 11. 10/14/74; N.L. Buoy (S); Depth 75'; HFVP 20". Evidence of erosion of fine sediment about the clay clump. Note vertical fissures of cohesive clay clump.
- Figure 12. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 12". Sediment surface features illustrating the cap of coarse sediment and shell fragments overlying a soft uncompacted mud-clay base. Epifaunal growth indicates current direction.
- Figure 13. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 4". Colonization of spoil surface by porifera and hydroids.
- Figure 14. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 6". Concentrations of annelid polychaete tubes were abundant and not limited to peripheral regions. Development appeared opportunistic and occurred in clonal clusters.
- Figure 15. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 6". Erosion of several clay mounds has occurred; burrowing activity has contributed to fracture and disintegration of these mounds.
- Figure 16. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 16". A high degree of burrowing was observed, primarily by the crabs, Cancer borealis and Cancer irroratus. Many clay banks were partially excavated by crabs and later expanded to larger burrow dimensions by the lobster.
- Figure 17. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 4". The entrance to the lobster burrow commonly noted on the newly deposited spoil materials. Burrow counts on certain transects were as high as 12-15.
- Figure 18. 9/14/74; N.L. Buoy (E); Depth 70'; HFVP 12". Loosely consolidated clay balls (.5-1 cm diameter) were observed to roll over the spoil pile. Dispersals of dredged material was not restricted solely to resuspension velocity but occurred often by this process.
- Figure 19. 5/10/76; N.L. Buoy (E); Depth 65'; HFVP 10". Concentration of Pitar and Mercenaria valves adjacent to clay clumps further indicate active erosion and sediment scouring.

H-7

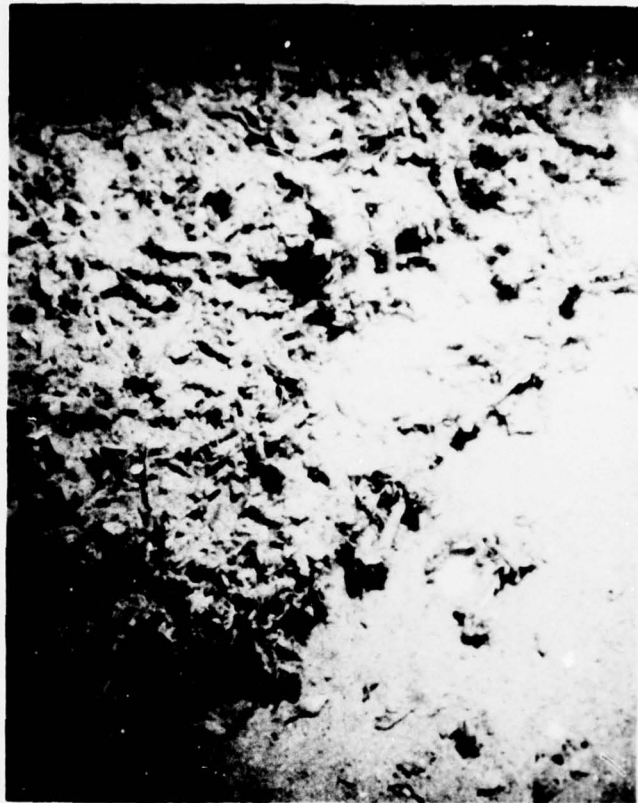


Figure 2



Figure 3



Figure 4



Figure 5



Figure 6

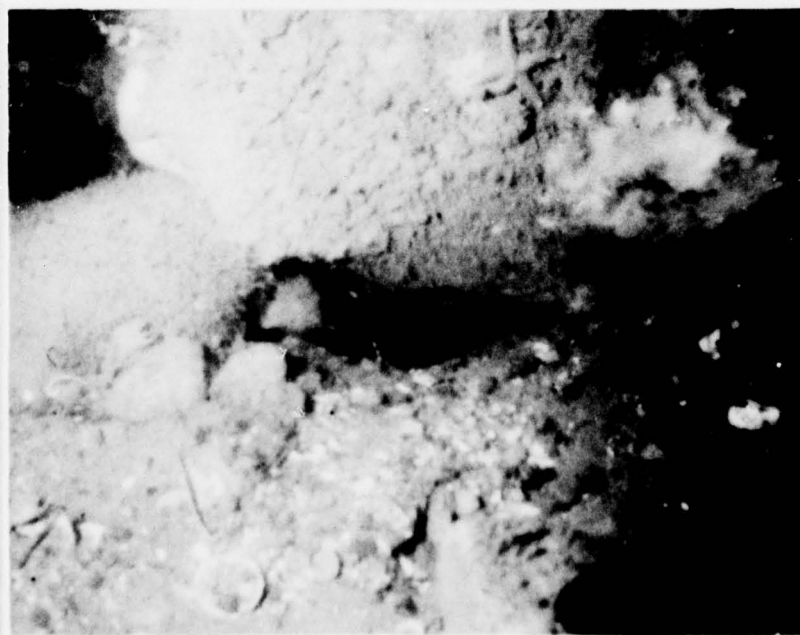


Figure 7

H-10

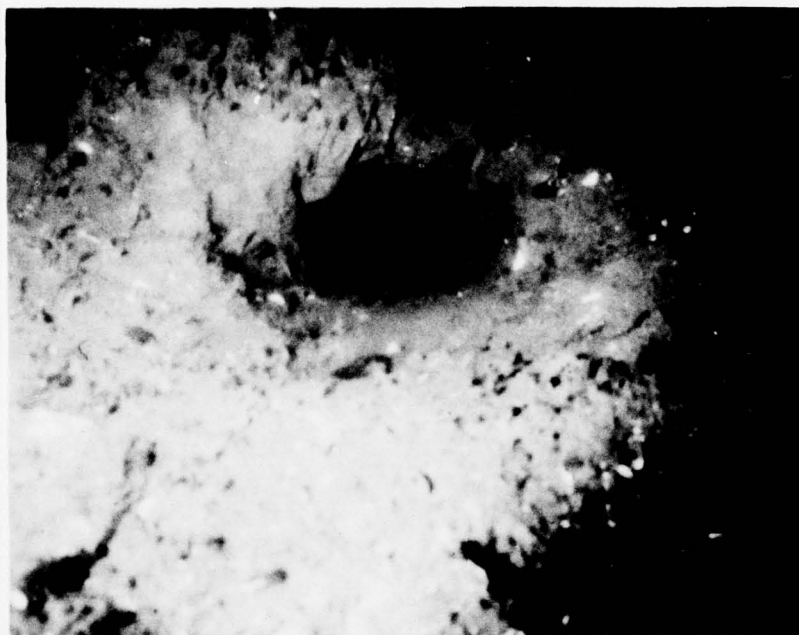


Figure 8



Figure 9

H-11



Figure 10

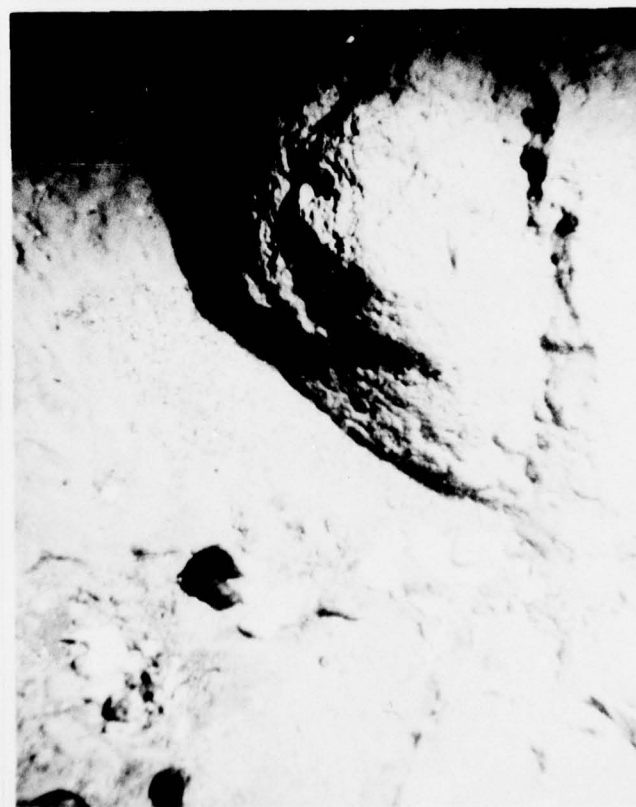


Figure 11

H-12



Figure 12

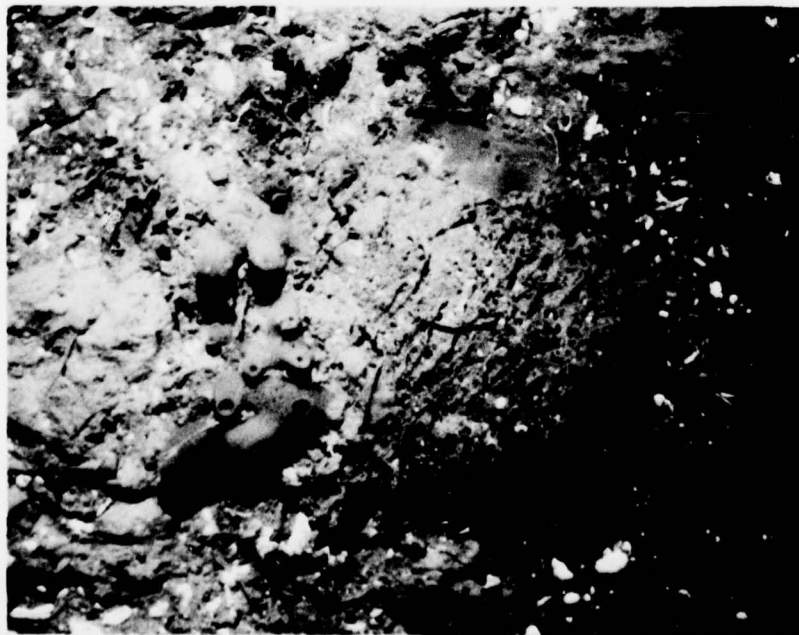


Figure 13

H-13

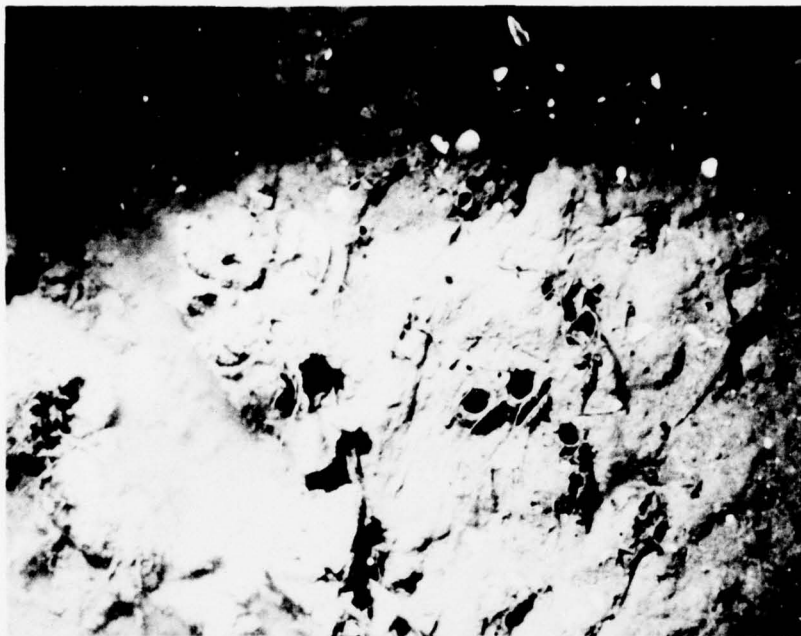


Figure 14



Figure 15

H-14

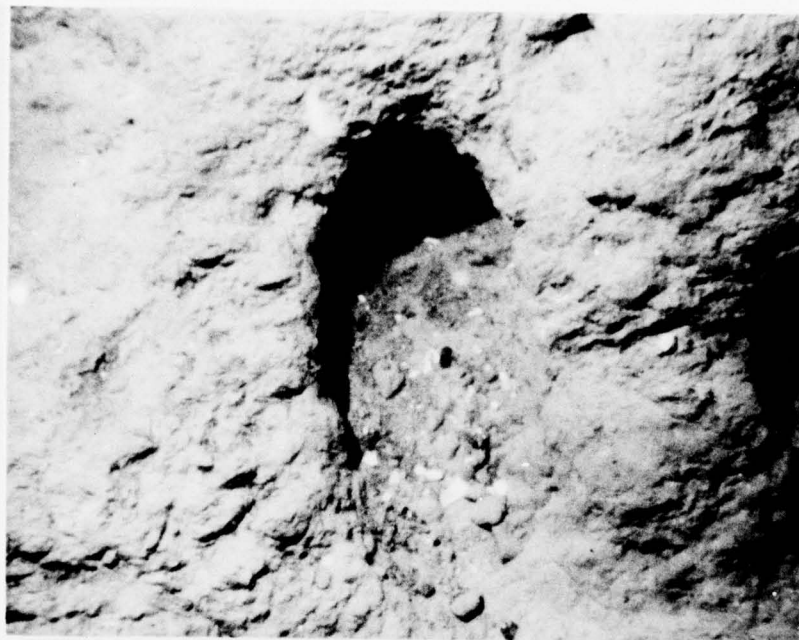


Figure 16



Figure 17

H-15

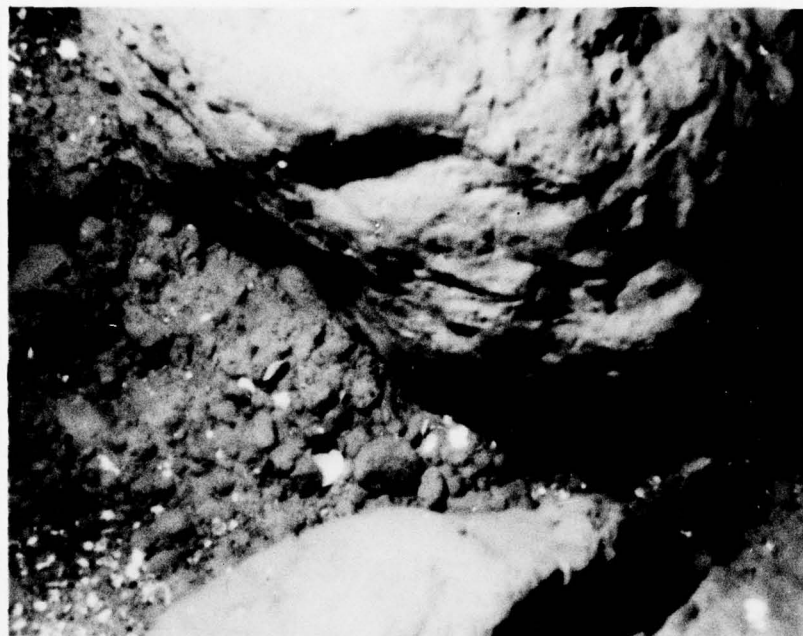


Figure 18



Figure 19

Lobster pot surveys were attempted to gauge relative fishing effort on the N.L. site during disposal. Three surveys (timed rectangular cruise patterns to cover the one n. mile square area with count of all pot buoys visible) were made on 6 August 1974, 27 September 1974, and 23 July 1975. Estimates of pot numbers were 56, 6, and 70 respectively. Pot number fluctuations were great and on certain dive survey cruises, no pots were observed within the disposal area. However, pots when present were concentrated in two locations; at the north central mound or in the southwest N.L. Buoy vicinity. On site interviews with lobstermen on the "Caroline J" and the "Debra Lynn", both Stonington boats, indicated no abnormalities in lobster condition or catch numbers. Both parties indicated the forty foot mound was traditionally fished while sets at the N.L. Buoy were experimental.

Existing features (May-June 1976, Figures 12-19) within a 50 m radius of the N.L. Buoy strongly suggest a gradual dispersal of spoil material has occurred. The immediate post-dredge topography has flattened considerably, distinct mounds and troughs are less obvious, and fine sediment has been eroded leaving 2-3 m diameter patch areas of coarse material. A high degree of spoil sediment compaction had occurred since the time of last survey (July 1975), and may have accounted in part to the leveling of previously noted bottom features.

Attempts to locate the northeast spoil border, one year after disposal had ceased were successful and a definite spoil-original bottom border could be detected. At depths less than 30 m, the mapping of a spoil periphery by underwater survey would be feasible. Permanent frontal locations could be marked by pingers to determine stability or advance with time. Also, the area of spoil coverage could be plotted by underwater circumnavigation of the spoil pile.

Summary:

1. Underwater photographs illustrate predisposal bottom conditions, lobster habitat on the site, characteristics of spoil after disposal, and evidence of biological recolonization.
2. Troughs and mounds of newly dumped material provided a relief to the normally flat bottom topography and was initially attractive to benthic fish and crustacea.
3. Survey of a north-central forty foot mound, a previous disposal site, indicated a high concentration of lobster, crabs, and flounder.
4. Observation of the water-spoil interface during 1.5 knots current conditions did not indicate visually noticable turbidity. The action of .5-1 centimeter diameter clay balls rolling over the spoil pile was observed as a sediment transport process.
5. Recent (May, June, 1976) dives indicated scouring of the sediment and disintegration of clay mounds has occurred. The general topography has flattened considerably. There is evidence of sorting, leaving a cap of shell fragments and 2-3 m diameter patches of coarse-gravel material. Spot dives indicate spoil spreading may have approached a point .5 n. miles southeast of the N.L. Buoy. A definite spoil-natural bottom border was detected in the northeast quadrant one year after disposal had ceased.